PROCEEDINGS

ACID MINE DRAINAGE

SEMINAR/WORKSHOP

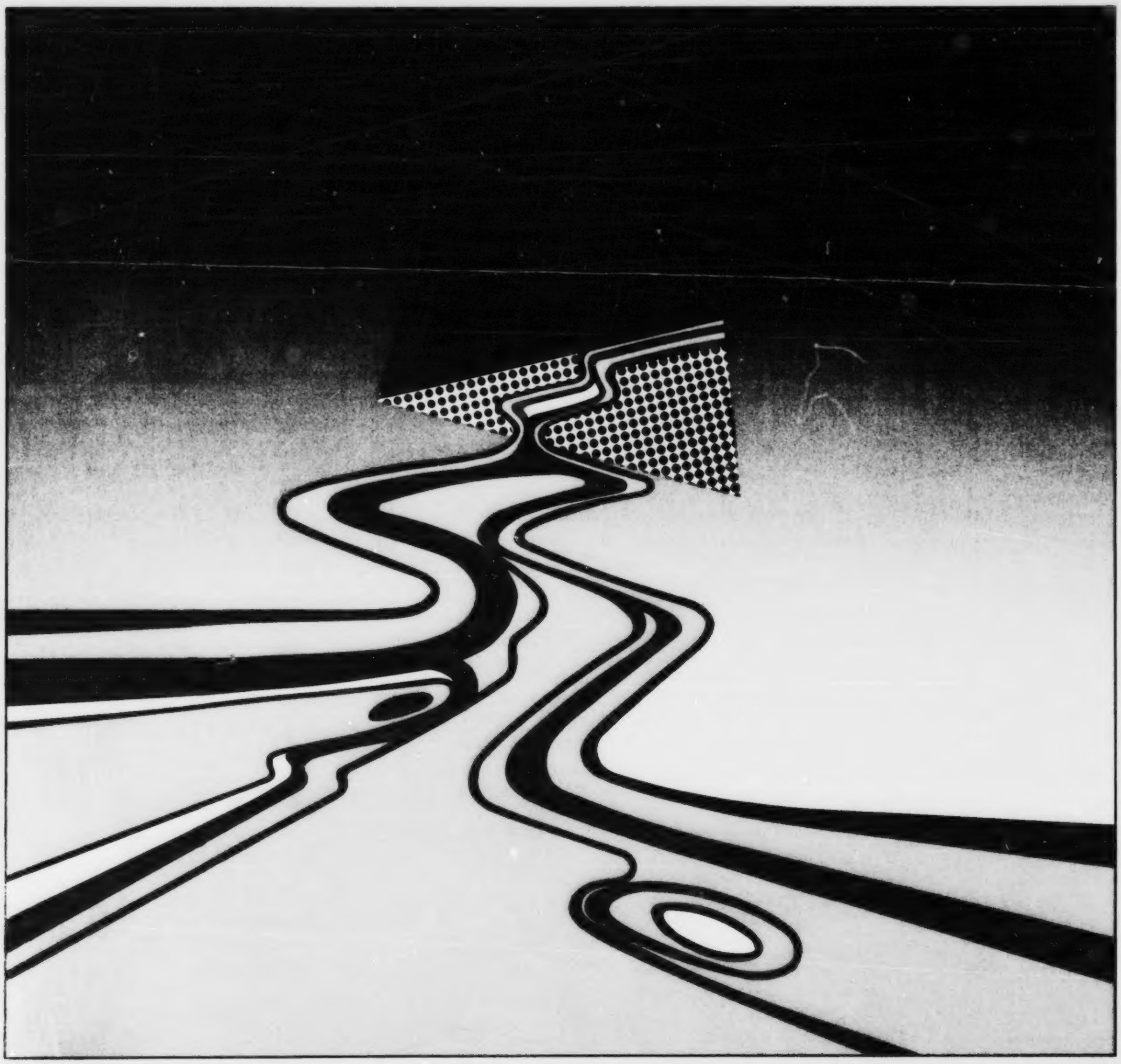
Halifax, Nova Scotia
23, 24, 25, 26 March, 1987

COMPTES RENDUS

DRAINAGE MINIER ACIDE

SÉMINAIRE/ATELIER

Halifax (Nouvelle-Écosse)
les 23, 24, 25, 26 mars, 1987



Environment Canada Environnement Canada

Canada

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Citadel Inn
Halifax, Nova Scotia

March 23-26, 1987

Conservation & Protection
Environment Canada
Ottawa, Ontario
K1A 0H3

Transport Canada
Airports Authority Group
Ottawa, Ontario
K1A 0N8

This Seminar/Workshop was supported by the

Panel for Energy Research and Development

COMPTES-RENDUS

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Catalogue No.
En 40-11-7/1987
ISBN 0-662-55005-6

N° de catalogue
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Ongoing acid mine drainage research activities are described and future research initiatives are also discussed.

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Bill Blakeman
Jim Scott
Mike Guilcher
Keith Ferguson

Transport Canada

John Worgan

Facilitators

OceanChem Group
Dartmouth, Nova Scotia

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- universités et centres de recherche;
- agences gouvernementales s'occupant de mines, de ressources naturelles, d'environnement, de transport et de développement économique;
- gouvernements locaux et régionaux.

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An Historic and Geographic Overview of Acid Mine Drainage

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Industrial Programs Branch
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Environment Canada
Ottawa, Ontario, K1A 0H3

Abstract:

Acid Mine Drainage (AMD) may be pragmatically defined as the inorganic chemical pollution resulting from mining activities. It is caused by the chemical and microbiological oxidation of sulphide containing minerals. Resulting drainage waters are usually of low pH and contain elevated levels of dissolved solids, total acidity, trace elements and inorganic compounds.

Although the generation of AMD can be traced to mining in antiquity, it has been a recognized mining related environmental phenomenon for the past 100 years and by the 1930's had become of significant environmental concern.

The main areas affected by AMD are historically the established coal mining districts of the Appalachians, especially the states of Pennsylvania and West Virginia. Several active and potential AMD generating sites exist throughout Canada primarily in Northern Ontario and British Columbia.

With further clarification of the main factors involved in the formation and propagation of AMD, achieved within the last 40 years, the chemical and microbiological mechanisms are now better understood. Improvements in physio-chemical wastewater treatment technologies, combined with the in-situ management of AMD, have resulted in a more complete methodology for AMD prevention, control and treatment.

INTRODUCTION

The use of minerals and metals by man as he evolved from the nomadic hunter-gatherer existence characteristic of the Paleolithic (Old Stone) Age and Neolithic (New Stone) Age to his modern form is nearly as old as his civilization.

From the early use of readily accessible bone, wood and stone materials, shaped and worked into tools or weapons, to the later use of metals and metallic ores, as represented by the Bronze Age (starting ca. 6000 BC) and Iron Age (Starting ca. 1000 BC) man has continued his ever expanding demand for metal products.

It is believed that gold, occurring as a native metal in placer deposits was the first metal used by man. Prized for its rarity, softness, colour and untarnished quality, gold was particularly useful for jewelry and as a standard of value (10). Copper also occurs in native form and being more abundant and harder than gold soon became of great value to early man as an implement or tool and as a weapon. This phase, referring to the initial use of pure copper is called the Chalcolithic (Copper-Stone) Age (1).

When copper is combined with tin, the resulting compound is an extremely hard and useful alloy called bronze. Bronze was an important alloy in the history of man and also to the Romans, whose empire at its zenith extended over most of present day Europe.

With the development and application of metallurgical skills, primitive man was no longer dependant upon metals occurring in their "native" or metallic form. Consequently, the deployment of complex, difficult to smelt ores was now within reach. The logical extension of this knowledge led to the widespread smelting of iron ore to produce metallic iron and the onset of the Iron Age which began at around 1000 B.C. Although the recovery of metallic iron from iron ore dates back to the fourteenth century BC, production in quantity had to wait several centuries.

The impact of these early mining and smelting activities on the air, water and land environments was probably negligible due to the extremely low production rates on the part of the "mines" and "smelters" and the very high assimilative capacity of the environment. One can only surmise, at this point, on the generation of acid mine drainage. However, if the ore mined contained sulphide (and given the fact that sulphur-oxidizing bacteria[1] had been in existence millions of years before the evolution of man), then in all probability acid mine drainage was produced and subsequently neutralized in the environment.

---

[1] The chemoautotrophic mode of life to which sulphur-oxidizing bacteria belong was discovered in 1890 (17).

In the area of fuel resources, there are no documented records of when or how mankind first discovered that a certain "black rock" would burn, although the use of coal (the "black rock") as a fuel is generally credited to the Chinese at about 1100 BC. Welsh Bronze Age cultures (which by nature of their location probably dates them later than the established date of the onset of the Bronze Age of 6000 BC) also used coal for funeral pyres.

With the institution of civilization and its consumptive requirements, increasing population growth led naturally to increasing demand for all types of resources (food, water, fuel (wood and coal), minerals and metals) and again one may surmise that technological developments kept pace with these increased demands. However, it is this increased demand that eventually lays the seeds for environmental contamination.

A history of the technological development (including metal and mineral mining) of man is too complex and lengthy a subject for the purposes of this paper. Instead, a summarized history of mining including the more pertinent dates is given in Table 1, with significant milestone dates noted (2).

By 950 BC, silver mines in southern Spain were operated by the Phoenicians, and it is here at about 750 BC that pyrite mining begins. May one then conclude that acid mine drainage was produced in this region? The probability is low, due to the generally dry climate of this area, however, the potential for acid mine drainage formation certainly existed.

By 122 AD, coal was used by the Romans during their occupation of the area now known as the United Kingdom. One can conclude this coal was obtained in all probability from surface outcroppings and mine drainage may have existed during wet periods although the prevailing climate of the United Kingdom during the period of Roman colonization (approximately 400 years) was amenable to grape cultivation. By 1238 AD, the first collieries were established in Newcastle, northern England, producing coal from pit and adit or drift mines.

By 1250 AD, Marco Polo had returned to Europe from China via the spice route and described the coal he saw in China as "stones which burn". Significantly, the Chinese had known of coal since 1100 BC, some 2,300 years before Marco Polo's visit. The first reference to coal in the new world is by 1300 AD, when the Hopi Indians of Arizona learned to burn the black rock found in sandstones. (Their use of coal has alternatly dated at 1000 AD).

By about 1250 AD, wood was becoming scarce in England and also rapidly increasing in price. The main city, London, began to import coal from surrounding towns. Consequently, the increased use of coal burning in London and the resultant air pollution problems, led King Edward I to proclaim that the burning of coal would be punishable by death[1]. It is not known how many people were executed for this crime, but five hundred years later Britain's coal reserves would fuel the Industrial Revolution, causing far more severe air pollution problems than during the time of King Edward I.

---

[1] In a general statement on North American air pollution, and revealing his knowledge of the subject, President Ronald (the Gipper) Reagan has remarked that trees are responsible for more pollution problems than coal burning thermal generating plants.

In 1556, the first edition of De Re Metallica was written by Georgius Agricola who, in one publication, summarized all available information relating to mining and smelting technology. Agricola observed contaminated waters arising from ore washing and of the subject wrote:

> ... "When the ores are washed, the water which has been used poisons the brooks and streams and either destroys the fish or drives them away" ...

This may have been the first written and recorded reference to the environmental impacts of acid mine drainage.

The capability to hoist men and ores from underground mines, using two oppositely driven water wheels, was developed in Hungary in 1475. This development is important in the history of mining because it would eventually allow the expansion of drift and shallow shaft mines to large scale undergruond mines. Surface and groundwater seeping into underground mines producing high pyritic sulphur coals, may generate large amounts of acid drainage.

Underground coal mining was given a further impetus in 1825 with the invention of the safety lamp by Sir Humphrey Davy. Explosions due to methane were greatly reduced or eliminated, permitting increased safety and productivity.

In their way, each of these inventions and discoveries, which are presented in summary form in Table 1, expanded the technology of mining and increased mining productivity. As a direct consequence of this, environmental disturbances were also noted - contamination of land, air and water. Acid mine drainage is but one aspect of this contamination.

COAL MINING IN AMERICA

Although America has vast coal resources historically the indigenous population was generally unaware of their existence or utility. With the exception of the Hopi Indians of Arizona, who as early as 1000-1300 AD used lignite for firing pottery, coal was never employed by the existing native population.

TABLE 1

AN ABBREVIATED WORLD HISTORY OF MINING AND MINERALS

41000 BC	-	Hematite (Fe_2O_3) mined for ritual paint in NW Swaziland.
25000 BC	-	Fired clay objects used in area now known as Czechoslavakia.
18000 BC	-	Fine leaf-shaped spearheads made from flint (chert) found in Spain and south France.
9500 BC	-	First old world metal artifact - a copper pendant found in the Shanidar Cave, N. Iraq.
5000 BC	-	Meteoric iron beads found near Cairo, Egypt.
4000 BC	-	A small earthen furnace located at Timna, Israel - the earliest extractive metallurgy discovered.
3500 BC	-	Gold vessels used in Ur in Mesopotamia (now Iraq).
3000 BC	-	Copper mined on the island of Cyprus.
	-	Remnants of copper found in Egyptian ruins.
2800 BC	-	Chinese legend and tradition suggest that Emperor Shen Nung discovered the process of smelting.
2000 BC	-	Earliest known metal workings in pre-Hispanic America evidenced in Gold foil found in Weycala in southern Peru.
1400 BC	-	Stone hammers and bronze chisels used in the Negev Mine, southern Israel, to obtain malachite ($CuCO_3.Cu(OH)_2$) copper ore.

IRON AGE STARTS 1000 BC

950 BC - Rio Tinto Silver Mines in southern Spain operated by the Phoenicians - pyrite mining started here about 750 BC.

122 - Coal used by romans in the area now known as United Kingdom.

1238 - First collieries established in Newcastle, England.

1250 - Marco Polo described the coal he saw in China as 'stones which burn' - he also brought back the first black powder formula from China.

1267 - First recorded reference to steel in England.

1300 - Hopi Indians learned to burn the black rock they found in Arizona sandstones.

1306 - In order to reduce air pollution in London, Edward I proclaimed the burning of coal punishable by death.

1350 - The blast furnace invented in Germany.

1400 - Alchemists (lead to gold) are generally credited with laying the foundations of modern metallurgy.

1524 - First recorded mining of copper by the Spaniards in Cuba

- Tin and copper mined in Taxco, Mexico.

1534 - Silver mines first worked in Taxco - here Cortez opened the first adit in the new world.

1556 - First ediction of De Re Metallica, the notable treatise on mining by Georgius Agricola.

1735 - Coke first successfully used to make iron by Abraham Darby in Shropshire, England.

1789 - Uranium discovered in Pitchblende by Martin Klaproth in Germany.

1815 - Sir Humphrey Davy invented the miners' safety lamp.

1825 - Introduction of monitors for hydraulic mining.

- Oersted produced small amounts of aluminum by chemical reaction.

1851 - First aerial photogrammetry from balloons and kites.

In the years 1607 to 1620 when the first British colonists established settlements in Virginia and Massachusetts Bay, Britain remained the main coal supplier to the colonies in the "new world". It was only a matter of time, however, until the coal reserves of America would be discovered, Britain would cease to be the main supplier and the "colonies" would gain their independence.

The first reference to coal in America dates to 1669 found, as legend has it, by a Virginian hunter pulling up a small tree along the river bank (Ref: Maryland Law School). The earliest official reference to coal in the United States is found in the account of Joliet's expedition to the Mississippi River in 1673. In 1680, LaSalle found coal near the Illinois River. Coal was discovered in Pennsylvania in 1698 by Gabriel Thomas and by 1736 maps of the Potomac River noted the presence of coal in Maryland and West Virginia. It was not until 1750 that coal was found in Ohio.

By about 1760, most of the colonies with coal knew of their reserves, but little had been done to use this resource. Wood was present in great abundance and Britain continued to pressure the colonies, during these pre-revolutionary times, to satisfy their limited requirements for coal by importing it from the "motherland".

The first actual coal production in the Appalachians occurred in 1768 in an area around what is now Pittsburg, on land that had been obtained from the Indians. By 1776, a year after the start of the American Revolution, coal was being used in the armory at Carlisle to help forge weapons for the Continental Army. After the Revolution, colonization proceeded westwards into western Pennsylvania, Virginia and West Virginia. Population growth in the eastern seaboard had by now put increasing demand on wood - which in turn resulted in greater reliance upon coal and further development of America's coal reserves. By 1830, coal production west of the Appalachian mountains had surpassed that to the east.

At this point in the development of the U.S. coal industry, which started in the Appalachian states and which are known to contain high sulphur coals, acid mine drainage had been produced and observed. One may also state that the environmental impacts were negligible due to the low coal production rates and comparatively high assimilative capacity of the environment.

Apart from Agricola's reference in De Re Metallica to contaminated ore processing waters that poisoned fish, it is believed by researchers on the subject (3) that the first reference to acid mine drainage in North America was made by Gabriel Thomas who in 1698 reported:

> "And I have reason to believe that there are good coals, also, for I observed the runs of water which have the same colour as that which proceeds from the coal mines in Wales".

Subsequent references to what may be interpreted as acid mine drainage were made in 1803 when T.M. Morris observed:

> "But the spring water issuing through fissures in the hills, which are only masses of coal, is so impregnated with bituminous and sulphurous particles as to be frequently nauseous to the taste and prejucidial to health",

and in 1809 when Joshua Gilpin noted:

> "Of the coal, several feet of a mixed kind of bad coal and iron abounding in sulphur and vitro efflorescing in white and yellow crystals".

These observations parallel the discovery and use of coal in North America which, when compared to the use of coal in Europe, is only slightly over 200 years. For Canada, and especially Nova Scotia, the use of coal is about 250 years, whereas for the remaining three Canadian coal producing provinces, Saskatchewan, Alberta and British Columbia, coal production dates to the early and mid-nineteenth century.

A summary of the history of coal production in North America and Canada is given in Tables 2 to 4.

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Nous croyons que les sujets abordés intéressent les personnes faisant partie de:

- compagnies minières exploitant les métaux, le charbon ou l'uranium;
- firmes de consultants;
- universités et centres de recherche;
- agences gouvernementales s'occupant de mines, de ressources naturelles, d'environnement, de transport et de développement économique;
- gouvernements locaux et régionaux.

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An Historic and Geographic Overview of Acid Mine Drainage

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Abstract:

Acid Mine Drainage (AMD) may be pragmatically defined as the inorganic chemical pollution resulting from mining activities. It is caused by the chemical and microbiological oxidation of sulphide containing minerals. Resulting drainage waters are usually of low pH and contain elevated levels of dissolved solids, total acidity, trace elements and inorganic compounds.

Although the generation of AMD can be traced to mining in antiquity, it has been a recognized mining related environmental phenomenon for the past 100 years and by the 1930's had become of significant environmental concern.

The main areas affected by AMD are historically the established coal mining districts of the Appalachians, especially the states of Pennsylvania and West Virginia. Several active and potential AMD generating sites exist throughout Canada primarily in Northern Ontario and British Columbia.

With further clarification of the main factors involved in the formation and propagation of AMD, achieved within the last 40 years, the chemical and microbiological mechanisms are now better understood. Improvements in physio-chemical wastewater treatment technologies, combined with the in-situ management of AMD, have resulted in a more complete methodology for AMD prevention, control and treatment.

INTRODUCTION

The use of minerals and metals by man as he evolved from the nomadic hunter-gatherer existence characteristic of the Paleolithic (Old Stone) Age and Neolithic (New Stone) Age to his modern form is nearly as old as his civilization.

From the early use of readily accessible bone, wood and stone materials, shaped and worked into tools or weapons, to the later use of metals and metallic ores, as represented by the Bronze Age (starting ca. 6000 BC) and Iron Age (Starting ca. 1000 BC) man has continued his ever expanding demand for metal products.

It is believed that gold, occurring as a native metal in placer deposits was the first metal used by man. Prized for its rarity, softness, colour and untarnished quality, gold was particularly useful for jewelry and as a standard of value (10). Copper also occurs in native form and being more abundant and harder than gold soon became of great value to early man as an implement or tool and as a weapon. This phase, referring to the initial use of pure copper is called the Chalcolithic (Copper-Stone) Age (1).

When copper is combined with tin, the resulting compound is an extremely hard and useful alloy called bronze. Bronze was an important alloy in the history of man and also to the Romans, whose empire at its zenith extended over most of present day Europe.

With the development and application of metallurgical skills, primitive man was no longer dependant upon metals occurring in their "native" or metallic form. Consequently, the deployment of complex, difficult to smelt ores was now within reach. The logical extension of this knowledge led to the widespread smelting of iron ore to produce metallic iron and the onset of the Iron Age which began at around 1000 B.C. Although the recovery of metallic iron from iron ore dates back to the fourteenth century BC, production in quantity had to wait several centuries.

The impact of these early mining and smelting activities on the air, water and land environments was probably negligible due to the extremely low production rates on the part of the "mines" and "smelters" and the very high assimilative capacity of the environment. One can only surmise, at this point, on the generation of acid mine drainage. However, if the ore mined contained sulphide (and given the fact that sulphur-oxidizing bacteria[1] had been in existence millions of years before the evolution of man), then in all probability acid mine drainage was produced and subsequently neutralized in the environment.

---

[1] The chemoautotrophic mode of life to which sulphur-oxidizing bacteria belong was discovered in 1890 (17).

In the area of fuel resources, there are no documented records of when or how mankind first discovered that a certain "black rock" would burn, although the use of coal (the "black rock") as a fuel is generally credited to the Chinese at about 1100 BC. Welsh Bronze Age cultures (which by nature of their location probably dates them later than the established date of the onset of the Bronze Age of 6000 BC) also used coal for funeral pyres.

With the institution of civilization and its consumptive requirements, increasing population growth led naturally to increasing demand for all types of resources (food, water, fuel (wood and coal), minerals and metals) and again one may surmise that technological developments kept pace with these increased demands. However, it is this increased demand that eventually lays the seeds for environmental contamination.

history of the technological development (including metal and mineral mining) of man is too complex and lengthy a subject for the purposes of this paper. Instead, a summarized history of mining including the more pertinent dates is given in Table 1, with significant milestone dates noted (2).

By 950 BC, silver mines in southern Spain were operated by the Phoenicians, and it is here at about 750 BC that pyrite mining begins. May one then conclude that acid mine drainage was produced in this region? The probability is low, due to the generally dry climate of this area, however, the potential for acid mine drainage formation certainly existed.

By 122 AD, coal was used by the Romans during their occupation of the area now known as the United Kingdom. One can conclude this coal was obtained in all probability from surface outcroppings and mine drainage may have existed during wet periods although the prevailing climate of the United Kingdom during the period of Roman colonization (approximately 400 years) was amenable to grape cultivation. By 1238 AD, the first collieries were established in Newcastle, northern England, producing coal from pit and adit or drift mines.

By 1250 AD, Marco Polo had returned to Europe from China via the spice route and described the coal he saw in China as "stones which burn". Significantly, the Chinese had known of coal since 1100 BC, some 2,300 years before Marco Polo's visit. The first reference to coal in the new world is by 1300 AD, when the Hopi Indians of Arizona learned to burn the black rock found in sandstones. (Their use of coal has alternatly dated at 1000 AD).

By about 1250 AD, wood was becoming scarce in England and also rapidly increasing in price. The main city, London, began to import coal from surrounding towns. Consequently, the increased use of coal burning in London and the resultant air pollution problems, led King Edward I to proclaim that the burning of coal would be punishable by death[1]. It is not known how many people were executed for this crime, but five hundred years later Britain's coal reserves would fuel the Industrial Revolution, causing far more severe air pollution problems than during the time of King Edward I.

---

[1] In a general statement on North American air pollution, and revealing his knowledge of the subject, President Ronald (the Gipper) Reagan has remarked that trees are responsible for more pollution problems than coal burning thermal generating plants.

In 1556, the first edition of De Re Metallica was written by Georgius Agricola who, in one publication, summarized all available information relating to mining and smelting technology. Agricola observed contaminated waters arising from ore washing and of the subject wrote:

> ... "When the ores are washed, the water which has been used poisons the brooks and streams and either destroys the fish or drives them away" ...

This may have been the first written and recorded reference to the environmental impacts of acid mine drainage.

The capability to hoist men and ores from underground mines, using two oppositely driven water wheels, was developed in Hungary in 1475. This development is important in the history of mining because it would eventually allow the expansion of drift and shallow shaft mines to large scale undergruond mines. Surface and groundwater seeping into underground mines producing high pyritic sulphur coals, may generate large amounts of acid drainage.

Underground coal mining was given a further impetus in 1825 with the invention of the safety lamp by Sir Humphrey Davy. Explosions due to methane were greatly reduced or eliminated, permitting increased safety and productivity.

In their way, each of these inventions and discoveries, which are presented in summary form in Table 1, expanded the technology of mining and increased mining productivity. As a direct consequence of this, environmental disturbances were also noted - contamination of land, air and water. Acid mine drainage is but one aspect of this contamination.

COAL MINING IN AMERICA

Although America has vast coal resources historically the indigenous population was generally unaware of their existence or utility. With the exception of the Hopi Indians of Arizona, who as early as 1000-1300 AD used lignite for firing pottery, coal was never employed by the existing native population.

TABLE 1

AN ABBREVIATED WORLD HISTORY OF MINING AND MINERALS

41000 BC - Hematite (Fe_2O_3) mined for ritual paint in NW Swaziland.

25000 BC - Fired clay objects used in area now known as Czechoslavakia.

18000 BC - Fine leaf-shaped spearheads made from flint (chert) found in Spain and south France.

9500 BC - First old world metal artifact - a copper pendant found in the Shanidar Cave, N. Iraq.

5000 BC - Meteoric iron beads found near Cairo, Egypt.

4000 BC - A small earthen furnace located at Timna, Israel - the earliest extractive metallurgy discovered.

3500 BC - Gold vessels used in Ur in Mesopotamia (now Iraq).

3000 BC - Copper mined on the island of Cyprus.
- Remnants of copper found in Egyptian ruins.

2800 BC - Chinese legend and tradition suggest that Emperor Shen Nung discovered the process of smelting.

2000 BC - Earliest known metal workings in pre-Hispanic America evidenced in Gold foil found in Weycala in southern Peru.

1400 BC - Stone hammers and bronze chisels used in the Negev Mine, southern Israel, to obtain malachite ($CuCO_3.Cu(OH)_2$) copper ore.

IRON AGE STARTS 1000 BC

950 BC - Rio Tinto Silver Mines in southern Spain operated by the Phoenicians - pyrite mining started here about 750 BC.

122 - Coal used by romans in the area now known as United Kingdom.

1238 - First collieries established in Newcastle, England.

1250 - Marco Polo described the coal he saw in China as 'stones which burn' - he also brought back the first black powder formula from China.

1267 - First recorded reference to steel in England.

1300 - Hopi Indians learned to burn the black rock they found in Arizona sandstones.

1306 - In order to reduce air pollution in London, Edward I proclaimed the burning of coal punishable by death.

1350 - The blast furnace invented in Germany.

1400 - Alchemists (lead to gold) are generally credited with laying the foundations of modern metallurgy.

1524 - First recorded mining of copper by the Spaniards in Cuba

- Tin and copper mined in Taxco, Mexico.

1534 - Silver mines first worked in Taxco - here Cortez opened the first adit in the new world.

1556 - First ediction of De Re Metallica, the notable treatise on mining by Georgius Agricola.

1735 - Coke first successfully used to make iron by Abraham Darby in Shropshire, England.

1789 - Uranium discovered in Pitchblende by Martin Klaproth in Germany.

1815 - Sir Humphrey Davy invented the miners' safety lamp.

1825 - Introduction of monitors for hydraulic mining.

- Oersted produced small amounts of aluminum by chemical reaction.

1851 - First aerial photogrammetry from balloons and kites.

In the years 1607 to 1620 when the first British colonists established settlements in Virginia and Massachusetts Bay, Britain remained the main coal supplier to the colonies in the "new world". It was only a matter of time, however, until the coal reserves of America would be discovered, Britain would cease to be the main supplier and the "colonies" would gain their independence.

The first reference to coal in America dates to 1669 found, as legend has it, by a Virginian hunter pulling up a small tree along the river bank (Ref: Maryland Law School). The earliest official reference to coal in the United States is found in the account of Joliet's expedition to the Mississippi River in 1673. In 1680, LaSalle found coal near the Illinois River. Coal was discovered in Pennsylvania in 1698 by Gabriel Thomas and by 1736 maps of the Potomac River noted the presence of coal in Maryland and West Virginia. It was not until 1750 that coal was found in Ohio.

By about 1760, most of the colonies with coal knew of their reserves, but little had been done to use this resource. Wood was present in great abundance and Britain continued to pressure the colonies, during these pre-revolutionary times, to satisfy their limited requirements for coal by importing it from the "motherland".

The first actual coal production in the Appalachians occurred in 1768 in an area around what is now Pittsburg, on land that had been obtained from the Indians. By 1776, a year after the start of the American Revolution, coal was being used in the armory at Carlisle to help forge weapons for the Continental Army. After the Revolution, colonization proceeded westwards into western Pennsylvania, Virginia and West Virginia. Population growth in the eastern seaboard had by now put increasing demand on wood - which in turn resulted in greater reliance upon coal and further development of America's coal reserves. By 1830, coal production west of the Appalachian mountains had surpassed that to the east.

At this point in the development of the U.S. coal industry, which started in the Appalachian states and which are known to contain high sulphur coals, acid mine drainage had been produced and observed. One may also state that the environmental impacts were negligible due to the low coal production rates and comparatively high assimilative capacity of the environment.

Apart from Agricola's reference in De Re Metallica to contaminated ore processing waters that poisoned fish, it is believed by researchers on the subject (3) that the first reference to acid mine drainage in North America was made by Gabriel Thomas who in 1698 reported:

> "And I have reason to believe that there are good coals, also, for I observed the runs of water which have the same colour as that which proceeds from the coal mines in Wales".

Subsequent references to what may be interpreted as acid mine drainage were made in 1803 when T.M. Morris observed:

> "But the spring water issuing through fissures in the hills, which are only masses of coal, is so impregnated with bituminous and sulphurous particles as to be frequently nauseous to the taste and prejucidial to health",

and in 1809 when Joshua Gilpin noted:

> "Of the coal, several feet of a mixed kind of bad coal and iron abounding in sulphur and vitro efflorescing in white and yellow crystals".

These observations parallel the discovery and use of coal in North America which, when compared to the use of coal in Europe, is only slightly over 200 years. For Canada, and especially Nova Scotia, the use of coal is about 250 years, whereas for the remaining three Canadian coal producing provinces, Saskatchewan, Alberta and British Columbia, coal production dates to the early and mid-nineteenth century.

A summary of the history of coal production in North America and Canada is given in Tables 2 to 4.

TABLE 2

COAL MINING IN U.S.

YEAR	LOCATION	EVENT
1669	Not known	First reference to stone coal in America
1673	Mississippi	Earliest official reference to coal in U.S.
1680	Not known	Coal found near Illinois River
1698	Pennsylvania	Discovery of coal noted by Gabriel Thomas
1736	W. Va., MD.	Presence of coal noted
1745	Not known	First coal mine established
1768	Appalachians	First actual production of coal

Legal problems of coal mine reclamation. The University of Maryland School of Law, March 1972.

TABLE 3

COAL MINING IN CANADA (NOVA SCOTIA)

YEAR	LOCATION	EVENT
1673	Cape Breton Island	Governor of Nova Scotia mentions existence of large coal seams on CBI
		Early mining activities by smugglers selling coal from outcroppings to New England states
1720	Cape Breton Island	First attempt at organized coal mining
		Coal supplied to markets on eastern seaboard of U.S.
1724	Cape Breton Island	First officially recorded export of coal from Cape Breton to Boston
1825	Sydney	General Mining Association formed and began systematic mining at Sydney area mines
1968	Nova Scotia	Formatin of CBDC; takes over operation of all mines

After (16)

TABLE 4

COAL MINING IN CANADA
(Saskatchewan, Alberta, British Columbia)

YEAR	LOCATION	EVENT
1857	Saskatchewan	Coal outcroppings on river banks used by Indians
1874	Saskatchewan	Mounted police detachments use local coal
1880's	Estevan (Sask.)	Coal shipped to Edmonton by barge
1841	Alberta	Blacksmiths at Fort Edmonton used coal for their forges
1870	Alberta	Early settlers and farmers obtain coal from outcroppings ("Gopher Holing")
1872	Lethbridge (Alberta)	First commercial mine started (drift mine into seam)
1835	Vancouver Island B.C.	First mine started on north end of Vancouver Island
1836-62	Central B.C.	Hudsons Bay Company mines from Susquash and Nanimo fields
1851	Namaimo (B.C.)	Nanaimo coal field production started

REF: Paine and Blakeman (16)

A HISTORY OF ACID MINE DRAINAGE RESEARCH

Although acid mine drainage had been observed as early as 1698 in Pennysl vania, the environmental impacts in this and other coal producing states during the early years of development were negligible. Consequently, during this early period there was no requirement for either AMD prevention or wastewater treatment. Coal production up to the early 1800's in the Appalachian states was by quarrying, a form of simplified strip mining. These small scale operations were conducted by hand in virtually every coal field opened before 1840. It is likely that, given the nature of Appalachian coal and climate of this region, acid mine drainage was produced and may have caused localized problems. For the Appalachian region as a whole, however, one may state that the generation of AMD remained well below the assimilative capacity of the environment.

Following the American civil war of 1861-1865 and the introduction of newer and heavier factory machines, coal was once again an important commodity very much in demand. Existing mining methods proved to be inadequate, however, and surface mines were soon exhausted. Coal companies were forced to go farther afield and farther underground.

In addition, development of the American railroads, from the 1840's-1860's, both as consumer and transporter of coal, resulted in further demands for coal. The railway was also a prime mover of coal to areas inaccessible by canal or river and thus assisted in developing and bringing industry to the American hinterland. In this leap-frogging fashion, industrial development stimulated by the railways resulted in further demand for coal, and increased coal production found new industrial markets.

At this point, two factors changed the fortunes of the coal companies and in doing so contributed directly to increasing the generation of acid mine drainage and the potential for acid mine drainage: underground mining and the widespread use of explosives.[1]

With coal once again in demand, new mines were established, surface mines were larger and underground mines were deeper. Further industrial development led to greater demands for coal with the result that the price of coal increased steadily through the 1870's and 1880's thereby spurring the demand for coal. This was satisfied by the establishment of new coal mines.

A direct consequence of expanded coal production in the Appalachian states was the formation of acid mine drainage and its impact on the environment. No longer was the environment, and specifically surface water systems, capable of assimilating increasing amounts of acid mine drainage. Acid mine drainage was rapidly becoming a coal-related environmental issue.

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[1] In 1866, "packaged powder", a combination of nitroglycerine and sawdust wrapped in cotton was introduced in the United States.

The first visual observations of the effects of acid mine drainage pollution were probably the red-yellow coloured precipitates of iron hydroxide and eradication of fish and other aquatic flora and fauna from surface water systems. The precipitate and its consequences became to be associated with and an unavoidable consequence of coal mining activities. Associated with the ferric hydroxide precipitate were a low pH, high acidity, high levels of dissolved trace elements and high sulphate concentration. Expanding coal production resuled in increased AMD generation and further contamination of surface water systems.

Concurrent with coal mining expansion and developments in mining technology were developments in chemistry, analytical chemistry (including instrumentation), water pollution control, domestic and industrial wastewater treatment and perhaps most importantly, a general awareness of the environment - an "environmental ethic". This combination of environmental awareness and science resulted in measures to understand the mechanism and reduce the formation of acid mine drainage.

A chronological history of the development of research undertaken to reduce AMD formation is useful for several reasons:

- to show how widespread the problem and is environmental impacts were;
- to show the development of understanding of the problem;
- to show the efficacy of early remedial measures.

Table 5 summarizes a personally selected history of AMD research (from abstracts (8)) conducted from 1910 to 1947. The year 1947 is important because it marks the discovery by Colmer and Hinkle (7) of the *Thiobacilli sp.* in acid mine drainage. These bacteria are now considered by researchers to be the catalyst and main causative agent in the generation of acid mine drainage (5).

TABLE 5

A HISTORY OF AMD RESEARCH FROM 1910-1948 (REF: COAL ABSTRACTS (GLEASON & RUSSELL)

YEAR	JOURNAL	PAPER	SUMMARY	
1910	Engineering Record 61 (16) 533-534	Germicidal effect of Mine Water and Farmers Wastes	. Effects of two wastes on sewage sewage bacteria	. Straight forward research paper establishing toxicity of AMD on sewage organisms
	Engineering Record 62, 371-372	The Acid Waters of Western Pennsylvania	. Effects and extent of acidity from mines on Pennsylvania rivers	. Documentation of acidity
1913	USBM Paper No. 58	Action of Acid Mine Water on the Insulation of Electric Conductors	. As indicated by title	. Studying the effects of AMD . Treating symptoms of AMD
1916	Coal Age 10 (8) 296-302	Chemistry in Coal Mining	. Functions of the chemist in coal mining . Analysis of several mine waters	. Initiation of chemists' role in study of AMD
1916	JAWWA 947-958	A New Raw Water Supply for the City of Mekeesport, Pennsylvania	. Changing supply from Youghiogheny to the Monongahela River due to AMD	. Shows impact of AMD . Treating symptoms not main cause of AMD
	Coal Age 10, 704-707	The Chemistry of Mine Water	. Wide variation of mine water . Lime and soda-ash discussed as neutralizing agents	. Characterization of AMD . Start of treatment technology
1917	Coal Age 11, 600-601	Use of Mine Water as Boiler Feed	. Mine water not recommended	
1918	Eng. Mining J. 106, 861	The Pollution of Streams	. Legal rights and responsibilities of mine owners	. Legal aspects of AMD now introduced
1921	JAWWA 8, 201-207	Pollution of River Water in the Pittsburg District	. Contributors of acidity to river water . Extent and character of AMD pollution	. Sources and characterization of AMD in a coal producing region

YEAR	JOURNAL	PAPER	SUMMARY	
1922	Ind. Eng. Chem. 14(2), 125-127	Nature of Acid Water From Coal Mines and the Determination of Acidity	. Composition of mine water . Methods for determining free and total acidity	. Improved characterization of AMD
1923	Eng. News-Record 91 (16), 638-641	Pollution of Water Suplies by Coal Mine Drainage	. Review of AMD . Relationship between coal mine drainage and rainfall	. Clarification of hydrological factors contributing to AMD
	Coal Age 23, 447-451	What Shall be Done about the Growing Evil of the Pollution of Streams by Mine Drainage	. Review of proposed laws to protect water supplies . Estimated that more than 1 million tons lime/year needed for neutralization of AMD to Ohio River	. Further involvement of legal aspects . Discussion of lime neutralization
1926	Mining Congress J. 12, 418-420	Relation of Drainage from Mines to Stream Pollution	. No known satisfactory solution to AMD problem . Neutralization costs: 34¢-57¢/ton . Softening costs: 70¢-$1.00/ton . Recommended careful investigation to problem before passage of legislation	. Establishment of lime neutralization and softening as main AMD treatment technology
1928	Mining Congress J. 14, 835-839	Observations of Acid Mine Drainage in Western Pennsylvania	. Water quality from working and abandoned mines and gob piles	. Still further characterization of AMD
	USBM RI 2892 8 p.	A Comparison of the Acidity of Waters from Some Active and Abandoned Mines	. Acidity and dissolved oxygen were lower in drainage from caved abandoned mines	. Concept of mine sealing as potential AMD prevention
1928	USPAT 1,685,300 Sept. 25, 1928	Proess and Apparatus for Treating Polluted Acid Wastes: Treatment of Polluted Waters	. Wastewater passed over a porous calcium carbonate . Additional treatment with lime	. Developments in neutralization technology
1930	Coal Age 35, 406-408	Wanted: More Research on Acid Mine Drainage	. Need for further study and field treatment work AMD	. Recognition of in-situ prevention of AMD
	Proc. Va. Acad. Sci. 4, 93-99	Developments in the Treatment of Acid Mine	. Neutralization, Dilution and Mine Sealing discussed as possible water treatment	. Further development in understanding AMD . Dilution would not, at present, be considered AMD treatment

YEAR	JOURNAL	PAPER	SUMMARY	
1932	USBM, R13146 15 pp.	Acidity from Drainage from High Pyritic Coal Areas in Pennsylvania	. Drainage from 39 coal mines sampled in 17 high pyritic areas in Pennsylvania	. Further comprehensive characterization of AMD from high pyritic areas
			. Drainage from high pyritic areas was more highly acid	
1933	W. Va. Univ., Eng. Sta. Ref: Bill-10, 38 pp.	Acid Mine Drainage from Bituminous Coal Mines	. Quantity of AMD correlated with coal vein, acreage mined and rainfall	. Hydrological aspects and idea of buffering
			. Experiments on buffering action performed	
	Ind. Eng. Chem. 25, 1355-1358	Oxidation of Pyritic Sulphur in Bituminous Coal	. Increasing temperature increased rate of pyrite oxidation (to sulphate)	. Becoming more scientific in approach to pyrite oxidation
			. Finer sizes of pyrite exhibit higher rates of oxidation	. With our present knowledge of microorganisms, these concepts are obvious
1935	Coal Age 40, 323-326	Sealing of Coal Mines-Will Reduce Acidity of their Effluent Waters	. Federal Mine Sealing Program	. Understanding that air/oxidation plays an important role in AMD formation
			. Types and efficacy of various seals used in program are discussed	
	JAWWA 27, 1186-1198	Acid Mine Drainage Control on Upper Ohio River Tributaries	. Assessment of economic damage by mine drainage in W. Va.	. First economic assessment
			. $1 million (estimated) annual loss to W. Va. by AMD	. Present worth would be far greater than $1 million
1936	The Pa. Eng. 12-13, 19 Oct. 1936	Pollution of Pittsbourgh's Rivers	. Contributors to water pollution in W. Pa. discussed	. Dilution of low river flows would not now be recommended
			. AMD counteracts effects of raw sewage	
			. Water storage reservoirs to dilute low flows are suggested	

YEAR	JOURNAL	PAPER	SUMMARY	
1936	W. Va. Univ. Tech. Bull. No. 8 p. 16-18	Unusual Difficulties in Treating Mine Water	. Mine discharge, after treatment, used as public water supply . Difficulties are taste, odour and high sulphate concentration	. Concept of using mine discharge after treatment as public water supply is unusual
1937	Ind. Eng. Chem. 29, 1040-1055	Pollution of Streams by Coal-Mine Drainage - Beneficial Effects of Sealing Abandoned Mines	. Reductions in acid production from 25 to over 80% by air sealing	. Oxiation route main acid generator
1938	Trans AIME 130, 425-444	Oxidation of Pyritic Sulphur in Coal Mines	. Investigation of the reaction mechanism of pyrite oxidation	. Experimental research on pyrite reaction
	W. Va. Univ. Eng. Expt. Sta., Ref: Bull. No. 18, 30 pp.	The Effects of Coal Mine Drainage on West Virginia Rivers and Water Supplies	. Air-sealing of abandoned mines, diversion of water from mines and construction of flood control dams will assure good stream conditions for various water users	. Concept of surface water diversion gaining favour.
1946	W. Va. Univ. Eng. Expt. Sta., Ind. Ref: Fellowship No. 11, Nov., 1946 14 pp.	Investigations on Coal Mine Drainage-Quarterly Report, July 1 to September 30, 1946	. Study of abandoned mine atmospheres, waste materials and organisms present in mines	. Consideration of organisms in mine water
	W. Va. Univ. Eng. Expt. Sta. Ind. Ref: Fellowship No. 11, July 1946	Investigations on Coal Mine Drainage - Semi-Annual Report for the First Half of 1946	. Development of a field method to analyze acid mine water for pH, total acidity, sulphuric acid, ferrous and ferric iron, aluminum, manganese and total sulphates	. Complete characterization of AMD for inorganic parameters . Microbiological analysis not included
1947	Science 106, 253-256 (Colmer & Hinkle)	The Role of Microorgaisms in Acid Mine Drainage	. Isolation of iron and sulphur oxidizing bacteria from AMD	. Most quoted paper indicating the bacterial role in AMD formation . This is a key paper in understanding AMD formation
1948	AIME Coal Div., Tech. Publ. 2381 5 pp. (Hinkle & Koehler)	The Action of Certain Microorganisms in Acid Mine Drainage	. Two organisms isolated from AMD . Believed to have a part in promoting the chemical reactions . One organism considered to be Thiobacillus thiooxidans	. Substantiation of earlier work (above)

History of Coal Mining in Canada

The colonization of Canada by the Europeans (French and British) dates approximately the same as that of America by the British. Early Canadian history revolved around the Hudsons Bay Company that was formed in Britain mainly to supply the British market with furs in exchange for trinkets and alcohol with the indigenous population who did the trapping.

The earliest mention of coal in Canada is in 1673 when Nicolas Denys, Governor of Nova Scotia mentioned in dispatches to France the existence of large coal seams on Cape Breton Island. Early mining activity in Nova Scotia from about 1673 to the early 1700's was conducted on a small scale by smugglers who obtained coal from cliff outcroppings and sold it to the New England States.[1]

In 1720, an attempt was made at organized coal mining to supply coal to the labour force constructing Fort Louisbourg. Subsequently, coal was supplied to the militia at Halifax and also to markets on the eastern seaboard of America. In the Sydney Coalfield, also in 1720, coal was mined by the French militia in the Port Morien district. Production and export of coal from Cape Breton Island increased, and in 1724 the first official export of coal from Cape Breton to Boston was recorded.

By 1825, the General Mining Association was formed and Systematic mining of the Sydney mines area had commenced. In 1968, the Cape Breton Development Corporation was formed and took over operation of all coal mines in Nova Scotia.

With expansion and development of Canada, coal was also found in Saskatchewan (lignite), Alberta (sub-bituminous and bituminous) and British Columbia (bituminous). Coal development in these provinces, however was later than that in Nova Scotia.

Table 3 presents a summary of the history of Coal Mining in Nova Scotia, and Table 7 presents a summary of the history of Coal Mining for Saskatchewan, Alberta and British Columbia.

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[1] Smuggling was possibly the only way to supply coal to the New England State who at that time purchased their coal from Britain.

TABLE 7 A HISTORY OF THE DEVELOPMENT OF COAL MINING ACTIVITIES IN SASKATCHEWAN, ALBERTA AND BRITISH COLUMBIA (16)

PROVINCE	DATE	LOCATION	COMMENTS
Saskatchewan	1857	-	It was reported that Indians were using coal outcroppings on river banks.
	1874	-	Coal dug for use by Mounted Police detachments.
	1880's	Estevan area	Mine production increases. Coal from Estavan area shipped by barge to Edmonton. Production is still small scale.
	1887	-	Coal Production = 400 t/y
	1890	-	200 t/y
	1900	-	40 000 t/y
	1910	-	181 000 t/y
	1920	-	335 000 t/y

Some low grade coal was mined in Southern Manitoba but there is no record of tonnages produced.

TABLE 7 A HISTORY OF THE DEVELOPMENT OF COAL MINING ACTIVITIES IN SASKATCHEWAN, ALBERTA AND BRITISH COLUMBIA (16)

PROVINCE	DATE	LOCATION	COMMENTS
Alberta	1841	Fort Edmonton	It was reported that blacksmiths at Fort Edmonton used coal for their forges. This coal outcropped on the river banks.
	1870	-	By 1870, early settlers and farmers were "gopher holing" i.e., digging into the coal seam from outcroppings.
	1872	Lethbridge	The first commercial mine was started by Nicholas Sherrin. This was a drift mine that followed into the seam after exposure. Coal from this mine supplied the mounties at Fort McLeod and was also hauled by ox-cart to Fort Benton, Montana. CPR opened another mine in the Lethbridge area (after 1872) and coal from this mine was sent by barge to Medicine Hat to supply the railways.
	1886		Coal Production = 40 000 t/y
	1890		128 000 t/y
	1900		311 000 t/y
	1910		2.9×10^6 t/y
	1920		6.9×10^6 t/y

TABLE 7 A HISTORY OF THE DEVELOPMENT OF COAL MINING ACTIVITIES IN SASKATCHEWAN, ALBERTA AND BRITISH COLUMBIA (16)

PROVINCE	DATE	LOCATION	COMMENTS
British Columbia	1835	Queen Charlotte Islands	Coal supplied ships on coastal routes.
	1835-36	Vancouver Island	The first mine started on the northern end of Vancouver Island providing coal for the coastal shipping services.
	1836-62	Central B.C.	Hudsons Bay Company mined a total of 65 000 t from the Susquash and Nanaimo fields.
	1851	Nanaimo	Nanaimo coal field production started.
	1900	Crows Nest Pass	The major development in B.C. coal mining took place at the start of the 20th century at the Crows Nest Pass. Coal supplied the railways.
	1886		Coal Production = 375 000 t/y
	1890		768 000 t/y
	1900		1.8×10^6 t/y
	1910		3.3×10^6 t/y
	1920		3.0×10^6 t/y depending on consumption by railways.

CHARACTERISTICS OF ACID MINE DRAINAGE

Although there is no typical or representative mine drainage, waters discharging from active and abandoned coal or base metal (sulphide ore) mines which create the major problem may be divided into four general categories as shown in Table 8.

The type of drainage induced depends upon the type of mine (surface or underground) and the nature of the geological surroundings. That is, the presence or absence of calcerous material capable of neutralizing acid drainage. Thus, the main characteristics of Class I acid drainage, drainage with the most severe environmental impacts, are how pH, high acidity and high sulphate levels.

Also present in acid mine drainage at relatively high concentrations are trace elements such as mercury, cadmium and lead derived from coal or base metal ores. The concentration, however, is element specific and depends upon the nature of ore or coal. Table 9 summarizes the ranges of component concentrations in Appalachian coal mine drainage and Table 10 presents acid mine drainage quality from acive base metal and abandoned uranium mines in Canada.

TABLE 8
MINE DRAINAGE CLASS (9)

Parameter	Class I Acid Discharge	Class II Partially Oxidized and/or Neutralized	Class III Oxidized and Neutralized and/or Alkaline	Class IV Neutralized and not Oxidized
pH	2 - 4.5	3.5 - 6.6	6.5 - 8.5	6.5 - 8.5
Acidity mg/L ($CaCo_3$)	1,000 - 15,000	0 - 1,000	0	0
Fe^{2+} mg/L	500 - 10,000	0 - 500	0	50 - 1,000
Fe^{3+} mg/L	0	0 - 1,000	0	0
Al mg/L	0 - 2,000	0 - 20	0	0
SO_4 mg/L	1,000 - 20,000	500 - 10,000	500 - 10,000	500 - 10,000

TABLE 9

RANGES OF COMPONENT CONCENTRATIONS IN APPALACHIAN COAL MINE DRAINAGE (4)

Major Components:	Suggested Ranges (mg/l)
Hydrogen ions	
pH	1.4 - 7.0
acidity, as $CaCO_3$	0 - 45,000
alkalinity, as $CaCO_3$	0 - 400
Soluble iron-total	1 - 10,000
Fe^{2+}	1 - 10,000
Fe^{3+}	1 - 1,000
Sulfate	1 - 20,000
Aluminum	1 - 2,000
Calcium	1 - 500
Magnesium	1 - 200
Total dissolved solids	5 - 10,000
Suspended solids	to 3,000
ferric oxyhydroxides, clays, silica and organics as: coal fines, sewage particles, plant components as algae.	
Minor Components:	
Sodium	to 100
Potassium	to 50
Silica	to 100
Chloride	to 200
Trace Components:	
Mn	to 50
Ni	to 5
V	to 2
Zn	to 10
Sr, Ba, Ti	to 5
Cd, Be, Cu, Ag, Co, Pb, Cr	1
Hg	0.05

TABLE 10
EXAMPLES OF ACID MINE DRAINAGE QUALITY (15)

Parameter[1]	Seepage from Abandoned Uranium Mine Tailings Pond in Ontario	Waste Rock Dump Seepage From Active Silver Mine in British Columbia	Minewater from Abandoned Underground Copper Mine in British Columbia
pH	2.0	2.8	3.5
Sulfate	7,440	7,650	1,500
Acidity	14,600	430	--
Iron	3,200	1,190	10.6
Manganese	5.6	78.3	6.4
Copper	3.6	89.8	16.5
Aluminum	588	359	--
Lead	0.67	2	0.1
Cadmium	0.05	0.5	0.143
Zinc	11.4	53.2	28.5
Arsenic	0.74	25	0.05
Nickel	3.2	8.0	0.06

[1] units are mg/l except pH

GEOGRAPHICAL EXTENT OF THE ACID MINE DRAINAGE PROBLEM

The oldest US coal mines are located in the Appalachian States of Pennsylvania, West Virginia, Ohio, Maryland, Tennessee and Kentucky, which have produced coal since the mid to late 18th century. The Appalachian States produce, primarily, a high sulphur bituminous coal and together account for approximately 25% of total US coal reserves. Of these states, the two main coal producers are Pennsylvania and West Virginia.

Most of the Appalachian coals are high pyrite coals and it is the biogeochemical oxidation of the ferrous disulphide (FeS_2) associated with the enclosing rock strata and also that present in the coal seam itself that causes the generation of mine drainage.

The land area affected by mining activity in the US exceeds 13 million acres (11). Of this total, 7 million have been undercut, 3 million disturbed by surface mining and 3 million given over to mining-related waste accumulations. It has been estimated that 10,000 mines of streams and 24,000 surface acres of impoundments and reservoirs are seriously affected by mine drainage. About 40% of this drainage comes from active mines, the remainder is from abandoned surface mines (25%) and shaft and drift mines (35%).

The significance of coal mining and coal mine drainage in the Appalachian region is that about 75% of the coal mine drainage in the United States occurs in this region (11). As noted earlier, the Appalachian States account for approximately 25% of total US coal reserves. A summary of the mine drainage problem is given in Table 11 which presents the coal production and stream mileage affected for the Appalachian States.

Table 11 The Acid Mine Drainage Problem in Appalachia (21)

State	Coal Production (1982) (tons)	Stream Mileage Affected (miles)
Pennsylvania	82,000,000	2,795
West Virginia[1]	-	1,100
Ohio	42,000,000	1,075
Maryland	2,500,000	450
Tennessee	11,500,000	994
Kentucky[1]	-	77(of 2100)[2]

[1] Coal production not available

[2] The low proportion of acid streams is apparently related to the presence of limestone in the coal-bearing rocks

The extent of the mine drainage pollution problem in Appalachia is best perceived graphically as shown in Figures 1 and 2 which present, respectively, pH and sulphuric acid loadings and sulphate concentrations in rivers in this region (11). The free mineral acid loads exceed 4,000 tons (of H_2SO_4) per day and are highest in the west Branch Susquehanna River, Monongahela River and Kiskiminetas River basis where the streams have pH values less than 4.5. Sulphate concentrations in excess of 50 ppm are found throughout the coal mining areas of the Appalachian region, the median concentration being 160 ppm in Pennsylvania and Ohio.

Canada, a producer of coal, several base metals, precious metals (gold, silver) and uranium, is not free from acid drainage. The primary sources of acid drainage in Canada are noted as follows:

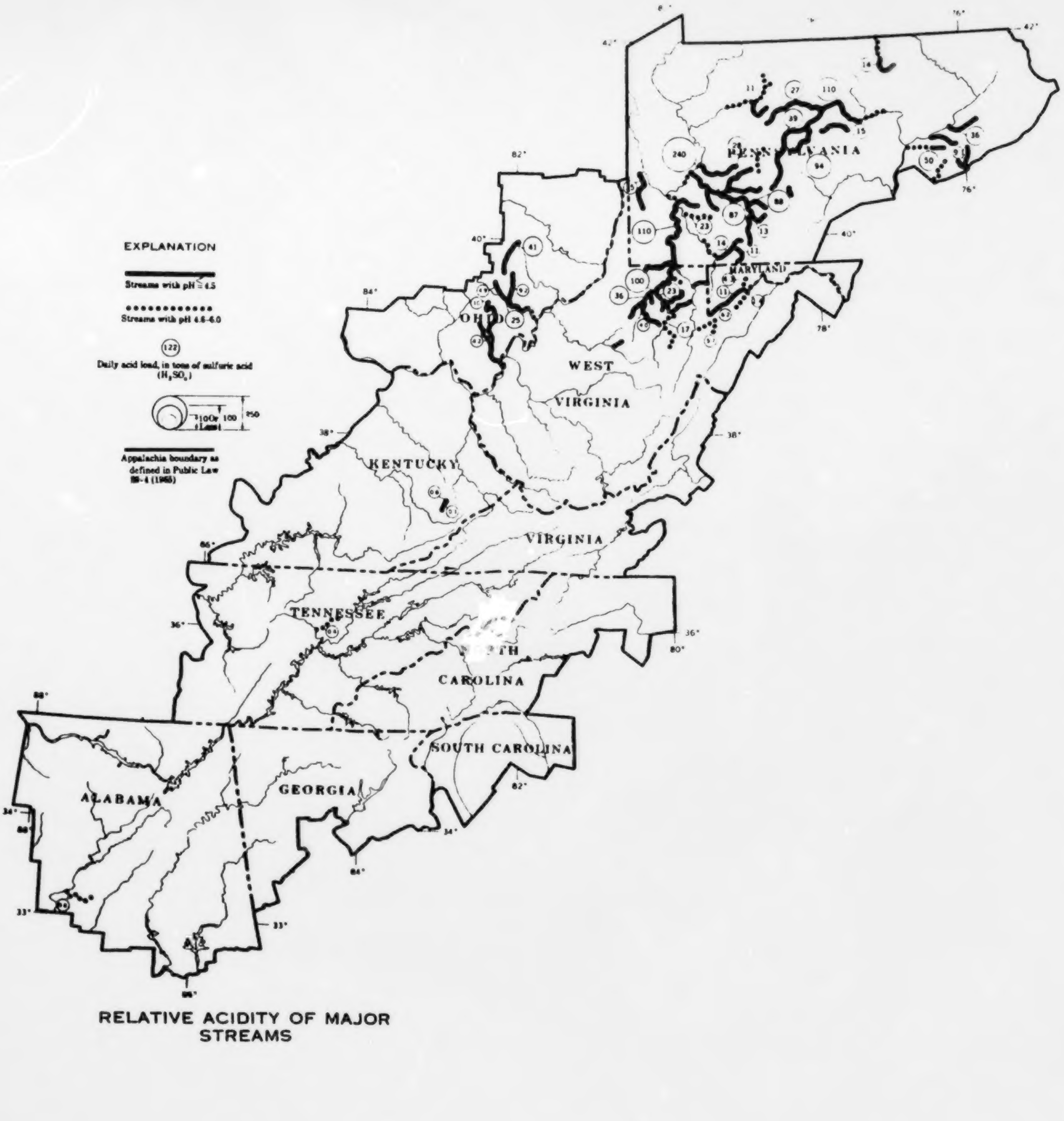
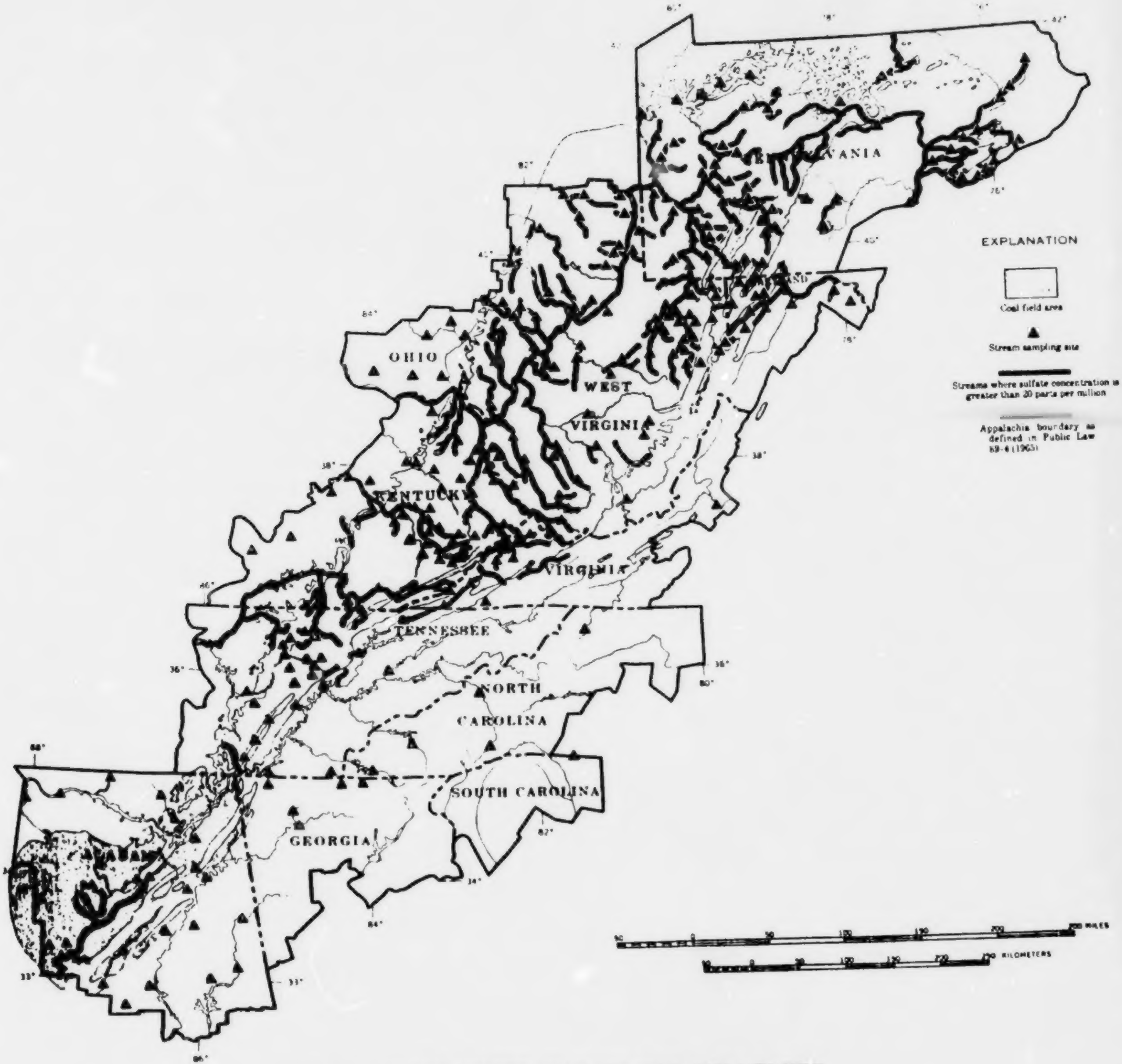

EXPLANATION
Streams with pH ≤ 4.5
Streams with pH 4.6-6.0
122
Daily acid load, in tons of sulfuric acid (H2SO4)
Appalachia boundary as defined in Public Law 89-4 (1965)
PENNSYLVANIA
OHIO
WEST VIRGINIA
MARYLAND
KENTUCKY
VIRGINIA
TENNESSEE
NORTH CAROLINA
SOUTH CAROLINA
GEORGIA
ALABAMA
42°
40°
38°
36°
34°
33°
76°
78°
80°
82°
84°
86°
88°
RELATIVE ACIDITY OF MAJOR STREAMS

Figure 1: Relative acidity of major streams in Appalachia


VANIA
OHIO
WEST
VIRGINIA
KENTUCKY
VIRGINIA
TENNESSEE
NORTH
CAROLINA
SOUTH CAROLINA
GEORGIA
76°
40°
38°
36°
80°
82°
84°
86°
88°
34°
33°
EXPLANATION
Coal field area
Stream sampling site
Streams where sulfate concentration is greater than 20 parts per million
Appalachia boundary as defined in Public Law 89-4 (1965)
50 0 50 100 150 200 MILES
50 0 50 100 150 200 250 KILOMETERS

STREAMS WHERE WATER QUALITY WAS INFLUENCED
BY COAL-MINE DRAINAGE

Figure 2: Sulphate concentrations in Appalachian streams

Coal: Coal is produced in the Atlantic Region (New Brunswick and Nova Scotia) and western Canada (Saskatchewan, Alberta and British Columbia). Atlantic coals are high pyritic sulphur coals, averaging 6-10% total sulphur for the Minto Coals of New Brunswick and up to 4% total sulphur for the Sydney coals of Nova Scotia. Consequently, generation of acid mine drainage in the Minto and Sydney coal fields is more of a problem. By comparison, Western Canadian Coals are very low sulphur coals, rarely exceeding about 0.8% total sulphur. Acid mine drainage is not a problem with these coals.

Base Metals: Base metals (such as lead, copper, nickel and zinc) are usually obtained from sulphide ores and are mined throughout the country. The main generator of acid drainage at base metal sites are sulphide tailings, the main waste product of mining.

Uranium: Uranium is produced from mines in Ontario (the Elliot Lake area) and Saskatchewan (Rabbit Lake, Cluff Lake areas). Of these two locations, the principle one is at Elliot Lake in Ontario which has many acid generating uranium tailings sites containing a total of about 100 million tons. A smaller acid generating site exists at Laredo in northern Saskatchewan, which from 1957 to 1960 operated a custom acid leach circuit mill. Approximately 360,000 tonnes of acid generating tailings were deposited at pH2 into adjacent Nero Lake.

Mining areas in Canada with potential acid drainage are presented in Figure 3.

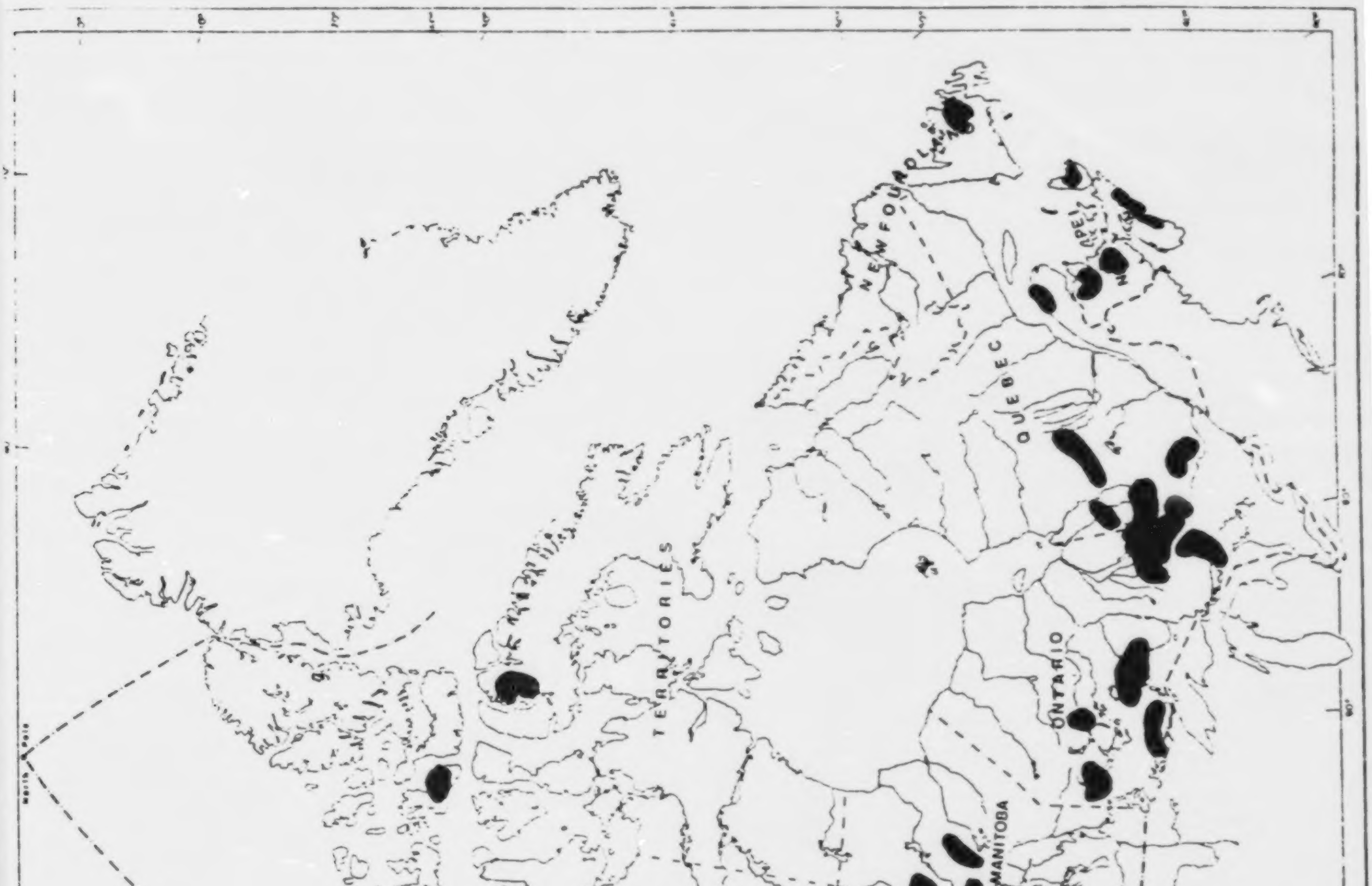

NEWFOUNDLAND
QUEBEC
ONTARIO
MANITOBA
SASKATCHEWAN
ALBERTA
BRITISH COLUMBIA
YUKON
NORTHWEST TERRITORIES
CANADA
Scale of miles
0 100 200 300 400

Figure 3: Potential Acid Drainage sites in Canada

Environmental Impacts of Acid Mine Drainage

The establishment and enforcement of surface water quality guidelines for elements and inorganic compounds in discharges to surface waters is primarily to safeguard aquatic life. For a given element or compound, standards exist such that the effluent concentrations of that element or compound is not beyond a specified value. This ensures that once in the receiving aquatic environment, and after allowing for dilution, the element or compound is maintained within a concentration range such that aquatic life is unharmed.

Acid mine drainage contains several elements and inorganic compounds at concentrations orders of magnitude greater than water quality guidelines established for prevention of damage to aquatic life. Several elements are either at lethal or threshold concentrations. In addition, there are synergistic interactions between certain metallic species such as zinc, copper and cadmiun that further compound the toxicity problem of these elements. Further, the toxicity of all elements is increased in acid (i.e. low pH) waters. Consequently, due to the complex chemical nature of acid mine drainage, it is difficult to assign toxicity to aquatic life to any single chemical constituent.

Table 12 compares the main components of acid mine drainage with water quality guidelines and Table 13 compares trace elements in acid mine drainage with water quality guidelines for those elements.

Acid mine drainage also eliminates sensitive aquatic life forms although more tolerant species occasionally flourish. The specialized flora and fauna of acid mine drainage in reality reflects harsh water quality conditions. For example fish are usually not found when the pH of a stream is lower than 4.5, whereas populations of midge larvae (Tendipes Sp.) may develop to nuisance proportions.

A qualitative biological examination of a stream contaminated by acid mine drainage (to pH 4.0 or lower) may reveal a community structure similar to the following:

Table 12 Concentration of the main components in Acid Mine Drainage with Water Quality Guidelines

Parameter [1]	Acid Mine Drainage [2]	Water Quality Guidelines [3]
Aluminum	1 - 2,000	0.1
Iron	1 - 10,000	0.3
Calcium	1 - 500	None
Magnesium	1 - 200	None
TDS	5 - 10,000	2,000 - 2,500
TSS	to 3,000	None
Sulphate	1 - 20,000	None
pH	1.4 - 7.0	6.5 - 9.5

Table 13 Concentration of Trace Elements in Acid Mine Drainage with Water Quality Guidelines

Element [1]	Acid Mine Drainage [2]	Water Quality Guidelines [3]
Manganese	to 50	0.02
Nickel	to 5	0.025
Vanadium	to 2	-
Zinc	to 10	0.03
Strontium	to 5	-
Barium	to 5	0.5
Titanium	to 5	-
Cadmium	L1	0.0002
Beryllium	L1	No guidelines
Copper	L1	0.0005
Silver	L1	No guidelines
Cobalt	L1	No guidelines
Lead	L1	0.03
Chromium	L1	0.05
Mercury	L0.05	0.0002

---

[1] All results mg/L

[2] Lovell, H. (4)

[3] Inland Waters Directorate (20)

Similar to the following:

(i) *Complex Plants*: Cattails (*Typha sp.*) and some mosses; other vascular life is generally not found in acid mine drainage.

(ii) *Algae*: Dense flowing mats of species of the green algae *Ulothrix* are common in gelatinous mats. Chlorophyll - containing flagellates (*Euglena* sp.) may colour stream beds dark green; microscopic examination commonly revels other species of green algae (*Microspora*, *Microthamnion*, *Chalmydomonas*) and diatoms.

(iii) *Benthic Invertebrates*: In severely contaminated stream reaches, such as near mine adits from which acid drainage emanates, no benthos will be found. In less severely polluted reaches, common inhabitants of streams, include midges, alderflies, fishflies, craneflies, dytiscid beetles and caddisfies.

(iv) *Fish*: At pH valves less than 4.5, fish are absent.

In general, the greater the contamination from acid mine drainage the less the number of species in a given community. Table 14 summarizes significant physical, chemical and biological criteria in evaluating mine drainage pollution of streams in Appalachia (14).

Table 14 - Physical, chemical, and biological Criteria significant in evaluating mine drainage pollution of streams in Appalachia.

Parameter	Range of Values of Concern	Major Water Use(s) Protected	Unpolluted Waters in Appalachia
1. pH	less than 6.0	uses involving aquatic life	6.0 - 9.0
2. Acidity	sufficient to lower alkalinity below 20 mg/l	uses involving aquatic life	less than alkalinity
3. Alkalinity	20 mg/l	uses involving aquatic life	20 mg/l
4. Sulfates	250 mg/l	domestic and industrial water supply	20 mg/l
5. Hardness	250 mg/l	domestic and industrial water supply	150 mg/l
6. Total iron	1.0 mg/l	uses involving aquatic life, domestic and industrial water supply	0.3 mg/l
7. Manganese	1.0 mg/l	uses involving aquatic life, domestic and industrial water supply	0.05 mg/l

Parameter	Range of Values of Concern	Major Water Use(s) Protected	Usual Values in Unpolluted Waters in Appalachia
8. Aluminum	0.5 mg/l	uses involving aquatic life	absent
9. Suspended solids	250 mg/l	uses involving aquatic life	100 mg/l (except during high runoff periods)
10. Dissolved solids	500 mg/l	domestic and industrial water supply	250 mg/l
11. Complex plants	Vascular plants community restricted; limited to *Typha* sp.; mosses present.	uses involving aquatic life	great variety of plant life
12. Simple plants	Algal community restricted; dominated by one to eight species, especially Ulothrix, Euglena, and pennate diatoms.	uses involving aquatic life	great variety of algae
13. Benthos	lacking in mayflies, stoneflies, most caldisfly species, crayfish and blackflies. *Tendipes* or *Sialis* *may be very abundant.*	uses involving aquatic life	great variety of benthos
14. Fishes	Lacking or much restricted.	fishing	varied fish fauna

The Thiobacilli: History and Characteristics

Although the *Thiobacillus sp* were first discovered to be present in acid mine drainage in 1947 by Colmer and Hinkle (7), the discovery of the *chemoautotrophic* mode of life in 1890 is credited to Sergei Winogradski who observed that H_2S could serve as electron donor in the respiration of *Beggiatoa sp*. (17). Winogradski writes:

> "Une systhèse complète de la matière organique par l'action d'être vivant peut s'accomplir sur notre planète indépendamment des rayons solaires".[1]

This observation envisioned an ecological niche in which a microbe derives all energy from the oxidation of inorganic sulphur species and all carbon from carbon dioxide, which in fact is the meaning of the word chemoautotrophic, as applied to microorganisms. Subsequent work by Nathanson on the *Beggiatoa sp.* led to the discovery and naming of the genus *Thiobacillus* in 1904 by Beijerinck, who is credited with discovering *Thiobacillus thioparus.*

The genus Thiobacilli are small (0.5 by 1.3 um), gram-negative, generally motile, rod-shaped bacteria which inhabit a wide variety of marine, freshwater and terrestrial environments. All thiobacilli are capable of autotrophic growth in simple salt media containing an oxidizable inorganic sulphur compound and carbon dioxide (18).

Facultative forms of thiobacilli, capable of adaptation to a heterotrophic environment have been found and there is increasing evidence that some organisms originally considered to be strict autotrophs are able to take up and utilize, to a limited extent, organic compounds. The majority of thiobacilli are strict aerobes but some may grow anaerobically with nitrate as an electron acceptor. One organism, *T. ferrooxidans*, can use the oxidation of ferrous iron as an alternative source of energy. Table 15 presents the nine species of Thiobacilli, their natural habitat and general characteristics.

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[1] The translation (19) is: "The complete synthesis of organic matter by the action of living organisms can be accomplished on our planet independently of sunlight".

Table 15 The Thiobacilli

The members of this genus are gram-negative, non-sporulating rods measuring 0.5 by 1-3u. Motile forms are polarly flagellated. All are capable of strictly autotrophic growth and most oxidize sulphide, elemental sulphur and thiosulphate to sulphate.

Organism	Habitat	General characteristics
T. thioparus	Canal water, mud, soil	Growth pH range 7.8-4.5 with an optinum near neutrality; generally aerobic and motile. Some strains grow anaerobically in the presence of nitrate.
T. neapolitanus (Thobacillus X)	Sea Water, corroding concrete structures	Strict aerobe with properties very similar to those of T. thioparus
T. denitrificans	Canal and river water, salt water, peat, composts, mud	Optimum growth pH near neutrality; oxidizes sulphur compounds anaerobically in the presence of nitrate; denitrifying ability lost on culturing in air; motile
T. thiooxidans	Soil	Optinum growth pH around 2; withstands 5% H_2SO_4; strict aerobe; motile
T. concretivorous	Corroding concrete structures	Very similar to T. thiooxidans
T. ferrooxidans	Acid mine and soil waters containing hydrogen sulphide	Strict aerobe; Optimum pH growth range 2.5-2.8; Also utilizes oxidation of ferrous iron and a source of energy; motile
T. novellus	Soils	Facultative autotroph; non-motile; optimum growth pH near neutrality

Organism	Habitat	General characteristics
T. intermedius	Fresh water mud	Facultative autotroph; growth pH range 2.0-7.0; motile; autotrophic growth stimulated in the presence of organic matter
T. thiocyanoxidans	Gas works liquor; Sewage effluent	Very similar to T. thioparus; thiocyanate oxidation serves as an energy source; oxidizes formate

After Trudinger (18).

FORMATION OF ACID MINE DRAINAGE

The main requirement in the formation of acid mine drainage in the presence of sulphur and sulphide minerals - without these minerals there can be no formation of acid mine drainage. Additional factors in AMD formation are oxygen (from air), water (H_2O in liquid or gaseous form) and a catalysing agent.

Early research on acid mine drainage in the coal mining areas of Appalachia (ca. 1920-1935) consisted of documenting the effects and extent of the problem (stream mileage affected, characteristics of drainage waters, impact of AMD on boilers, etc.) rather than the fundamentals causes. From this research, AMD prevention measures were proposed which generally consisted of air sealing of abandoned mines (to limit air ingress) and diversion of surface water from the mine. (Additional information on early research activities conducted on acid mine drainage may be obtained from Gleason and Russell (8)).

In 1947, a milestone paper by Colmer and Hinkle (7) showed the presence of T. ferrooxidans in mine drainage water and hinted at the possibility that this bacteria may be the case of AMD. Once discovered to be present, microbiological investigation of acid mine drainage waters expanded and in 1953 experiments conducted by Leathen et al (5,8) suggested that T. ferrooxidans increased the rate of pyrite oxidation.

In 1970, Singer and Stumm (12) showed that abiotic oxidation of pyrite (that is, the direct oxidation of pyrite by air) was too slow to generate the amount of acidity observed in nature. Another stronger, faster acting oxidizing agent is required. This agent is now generally regarded to be Thiobacillus ferrooxidans.

The main reactions responsible for pyrite oxidation and ultimately formation of acid mine drainage, are summarized in Table 16. According to Kleinmann (5), who provides an excellent description of the three stages of the chemistry of pyrite oxidation, the actual reaction process occurs in a multistage sequence dependent upon the activity of T. ferrooxidans and solution Eh and pH.

Table 16 Reactions Responsible for Pyrite Oxidation

1. $FeS_2 + 7/2\ O_2 + H_2O \longrightarrow Fe^{2+} + 2SO_4{}^{2-} + 2H^+$
2. $Fe^{2+} + 5/2\ H_2O + \frac{1}{4}O_2 \longrightarrow Fe(OH)_3(S) + 2H^+$
3. $Fe^{2+} + \frac{1}{4}\ O_2 + H^+ \longrightarrow Fe^{3+} + \frac{1}{2}\ H_2O$
4. $FeS_2 + 14\ Fe^{3+} + 8H_2O \longrightarrow 15Fe^{2+} + 2SO_4{}^{2-} + 16H^+$

Stage 1
Reaction 1: proceeds both abiotically and by direct bacterial oxidation.
Reaction 2: proceeds abiotically, slows down as pH falls.

In this stage: pH above approx. 4.5; high SO_4; low iron; little or no acidity.

Stage 2
Reaction 1: proceeds abiotically and by direct bacterial oxidation
Reaction 2: proceeds at rate determined primarily by activity of *T. ferroxidans*

In this stage: approx. pH range of 2.5-4.5; high SO_4; acidity and total iron inreasing; low Fe^{3+}/Fe^{2+} ratio.

Stage 3
Reaction 3: proceeds at rate totally determined by activity of *T. ferroxidans*.

Reaction 4: proceeds at rate primarily determined by rate of reaction 3.

In this stage: pH below approximately 2.5; high SO_4, acidity, total iron and Fe^{3+}/Fe^{2+} ratio.

CONCLUSIONS

(1) Acid Mine Drainage (AMD) is not a new problem, and historically, can be traced to the mining of pyrite in antiquity (ca. 750 BC) by the Phoeneicians in Southern Spain.

Giorgius Agricola in De Re Metallica (1556) writes:

> "When the ores are washed the water which has been used, poisons the brooks and streams and either destroys the fish or drives them away"

(2) In North America, the coal mining areas of the Appalachian states, and principally Pennsylvania and West Virginia, have suffered from acid drainage produced by extensive surface and underground coal mining.

(3) In Canada, there are several potential acid drainage sites. These sites may be found at coal (Nova Scotia, New Brunswick), base metal (nation wide) and uranium (Ontario, Saskatchewan) mining sites. For base metal and uranium sites, acid drainage is produced from sulphide tailings.

(4) Although acid mine drainage had been recognized in the US since the late 17th century, early research (ca. 1910-1935) into its control centered on characterization and extent of the problem. Control of AMD was to be achieved by mine sealing and surface water diversion. The main turning point in the understanding of AMD formation was in 1947 with the discovery of *Thiobacillus ferrooxidans* in acid drainage from bituminous coal mines.

(5) Historically, the chemoautotrophic mode of life, to which all species of Thiobacilli belong, had been discovered in 1890 by Winogradsky. The Thiobacilli were subsequently isolated in 1902 and named in 1904.

(6) The primary requirements for acid mine drainage are sulphide minerals, air (O_2), water and bacteria.

(7) The basic stoichiometric chemistry of acid mine drainage formation (i.e. chemical reactions) have been well documented and known of for several years. Reduced to simplest fundamentals, the chemistry involves:

oxidation of ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}) by bacteria

$$Fe^{2+} + \text{bacteria} \longrightarrow Fe^{3+} \qquad (1)$$

then, oxidation of sulphide minerals by bacterially produced ferric iron to regenerate ferrous iron (which becomes available for bacterial oxidation) and produce acid drainage.

$$Fe^{3+} + \text{pyrite} \rightarrow Fe^{2+} + \text{AMD} \qquad (2)$$

These two reactions operate in a cyclical fashion and depend on each other. Reaction (1) is the rate limiting step.

(8) The main characteristics of acid mine drainage are trace elements (primarily Al, Fe but also Cu, Zn, Ni, etc.), high sulphate concentrations (up to 10,000 mg/L), high total acidity (10,000 mg/L as $CaCO_3$) and low pH (at worst case, around pH 2).

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THE BIOGEOCHEMISTRY OF ACID GENERATION IN SULPHIDE TAILINGS AND WASTE ROCK

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ABSTRACT

Acid generation is perhaps the single greatest concern associated with the decommissioning and reclamation of sulphidic tailings and waste rock areas in Canada. This paper reviews the primary biochemical reactions for sulphide oxidation and addresses how these reactions affect the chemistry of runoff and seepage from these waste impoundments as the oxidation proceeds through time. A state-of-the art review of the primary kinetic factors which can modify the biological and chemical oxidation rates is presented. The key biochemical parameters affecting acid generation are temperature, pH, oxygen concentration, carbon dioxide concentration, the presence of nutrients, moisture content, the concentration of potential inhibitors and mass and surface area of pyrite present. Reference is made to the recently completed RATAP acid generation model prepared by SENES Consultants and Beak Consultants on behalf of the National Uranium Tailings Program Office.

1.0 INTRODUCTION

The interdependency of geochemical, physical and biological factors which control the rate and quantity of sulphide oxidation in tailings and waste rock is not clearly understood. These factors include temperature, bacterial population density, the oxygen profile, the porewater pH, mineralogical composition of the solids, and the mass and effective grain size of the sulphide particles. To model these effects, one must fully understand the biogeochemistry of the sulphide oxidation as it relates to both the biological and chemical oxidation of sulphide minerals. For simplicity, pyrite is used as a surrogate for sulphide minerals as this is the most common form of sulphide mineralization in Canadian mine tailings and waste rock.

The overall stoichiometric equations governing the oxidation of pyrite and precipitation of reaction products are most commonly written as:

$$2FeS_2 + 7O_2 + 2H_2O \rightleftharpoons 2FeSO_4 + 2H_2SO_4 \qquad \text{(Eq.1)}$$

$$2FeSO_4 + .5O_2 + H_2SO_4 \rightleftharpoons Fe_2(SO_4)_3 + H_2O \qquad \text{(Eq.2a)}$$

$$FeSO_4 + Ca(OH)_2 \rightleftharpoons Fe(OH)_2 + CaSO_4 \qquad \text{(Eq.2b)}$$

$$Fe(OH)_2 + O_2 \rightleftharpoons Fe(OH)_3 \qquad \text{(Eq.2c)}$$

$$Fe_2(SO_4)_3 + 6H_2O \rightleftharpoons 2Fe(OH)_3 + 3H_2SO_4 \qquad \text{(Eq.3)}$$

$$FeS_2 + 7Fe_2(SO_4)_3 + 8H_2O \rightleftharpoons 15FeSO_4 + 8H_2SO_4 \qquad \text{(Eq.4)}$$

$$2FeS_2 + 7.5O_2 + 7H_2O \rightleftharpoons 2Fe(OH)_3 + 4H_2SO_4 \qquad \text{(Eq.5)}$$

These reactions are simplications of a complex set of reactions that may well have numerous intermediate steps and half reactions involving the formation of peroxide, various sulphur compounds including sulphite, elemental sulphur and thiosalts.

Equation 5 is a stoichiometric summary of the pyrite oxidation reactions leading to ferric hydroxide precipitation and sulphuric acid production. Under the complete oxidation of pyrite by biological or chemical processes, one mole of pyrite produces 2 moles of sulphuric acid. The reaction can be broken down into 2 parts, one involving the oxidation of pyrite to form ferrous ion and the second,the oxidation of ferrous ion to ferric ion. The overall oxidation of pyrite is strongly pH dependant. This dependence results from the chemistry of iron and optimal pH conditions for microbial oxidation. Under alkaline conditions, chemical oxidation dominates and pyrite oxidation proceeds slowly.

Where buffering conditions are sufficient to prevent acid pH levels from forming, ferric ion is precipitated and sulphuric acid is consumed by buffering minerals such as calcite. In the presence of oxygen at neutral to alkaline pH levels, the ferrous ion is converted to the ferric ion and precipitated. The oxidation of ferrous ion is often considered the rate limiting step in chemical oxidation.

As the pH decreases into the acidic range biological oxidation becomes more prominent. The oxidation of ferrous ion to ferric ion is greatly accelerated by the microbial activity. Microbially assisted pyrite oxidation has been reported to be several orders of magnitude greater than chemical oxidation under optimal conditions. In addition, at low pH conditions ferric ion solubility increases, thus increasing the availability of ferric ion as an oxidant.

As the pH drops below 2.0, however, chemical oxidation may again become the primary pyrite oxidation mechanism. Results of acid generation model RATAP (SENES *et al.*, 1986) indicate that chemical oxidation is the prime mechanism at these low pH values.

Under anoxic conditions ferric ion will oxidize pyrite forming ferrous ion (Eq. 4). This reaction commonly occurs in tailings and waste rock areas below the water tables or in areas where oxygen is depleted. Therefore, acid runoff may contain hundreds or thousands of mg/L of ferrous ion which upon neutralization (Eq. 2b, 2c) will precipitate as ferrous or ferric hydroxide.

2.0 BACTERIAL OXIDATION AND CHEMICAL OXIDATION KINETICS

The oxidation of pyritic tailings in the presence of bacteria, principally the group of T. ferroxidans is caused by a sequence of chemical reactions. Bacterially-assisted oxidation of pyrite involves both direct and indirect mechanisms. Direct mechanisms involve attack of the pyrite surface by bacteria while indirect mechanisms include the dissolution of pyrite by a ferric ion in solution and the oxidation of ferrous ion by the bacteria. Chemical oxidation involves only the oxidation of pyrite by chemical mechanisms and the subsequent oxidation of the ferrous ion.

The rate of bacterially-assisted pyrite oxidation, R_B, is a function of several biological and environmental parameters (Hoffmann et al., 1981; Halbert et al., 1983; Jaynes et al., 1984; SENES, 1984). It may be stated by the following expression.

$$R_B = f(X_p, Y_{x/s}, pH, T, P_{O2}, P_{CO2}, S, [N], [P], E_a, A, [I])$$

where

R_B = rate of microbial oxidation

X_p = bacterial population density

$Y_{x/s}$ = growth yield coefficient

pH = measure of hydronium concentration

T = soil temperature

P_{O2} = oxygen partial pressure

P_{CO2} = carbon dioxide partial pressure

S = degree of saturation

[N] = nitrate or ammonia concentration

[P] = phosphorus concentration

[I] = inhibitor concentration

E_a = biological energy of activation

A = surface area of pyrite

As with the biological rate, chemical oxidation rate, R_c, is a function of several environmental parameters:

$$R_c = f\ (pH, T, E_c, A, P_{O2}, S, [Fe^{3+}])$$

where

R_c = rate of chemical oxidation

$[Fe^{3+}]$ = concentration of the ferric ion

E_c = chemical energy of activation

Therefore, the total rate of oxidation $R = R_B + R_c$. SENES (1986) calibrated the relative significance of chemical and biological reactions for pyrite oxidation in Elliot Lake tailings. At optimal laboratory conditions (pH = 3.25, 30°C, 100 per cent saturation, O_2 = .21 atm), the biological oxidation reaction rate was 100 times magnitude greater than the chemical oxidation rate. However, under optimal field conditions, this rate is lowered to about 20 times chemical oxidation rates. These optimal conditions would occur for short periods of time and in very selected areas of a tailings pile or waste rock dump. To illustrate this phenomenum, Figure 1.0 presents a plot of the rate of pyrite oxidation as a function of depth for years one, four and ten during the month of June for an Elliot Lake type tailings. During years one and ten the profiles for the rates of pyrite oxidation are very similar. It is during these periods that chemical oxidation controls. In year one, calcite is present in all layers and the pH of the tailings porewater is buffered at about 6.4 to 6.6. In year ten only ferric hydroxide is present in all the layers and the pH is buffered at about 1.7. The bacterial activity in both cases is low because of the unfavourable pH conditions. In contrast, during year four when the solid phase system buffers the pH at about 3.5, bacterial oxidation is the controlling factor. As suggested by the curves on Figure 1.0 the rate of pyrite oxidation per unit volume of tailings is about 6 times greater when bacteria-assisted oxidation controls than when chemical oxidation controls.

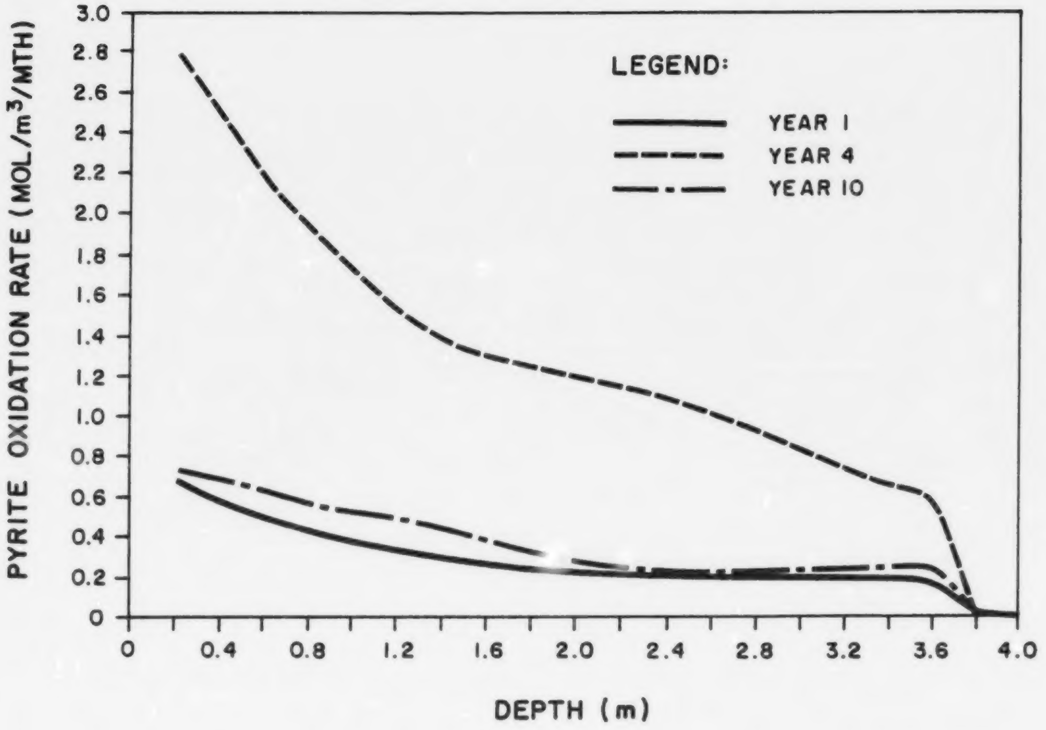


FIGURE 1

RATE OF PYRITE OXIDATION vs. DEPTH FOR THE MONTH OF JUNE

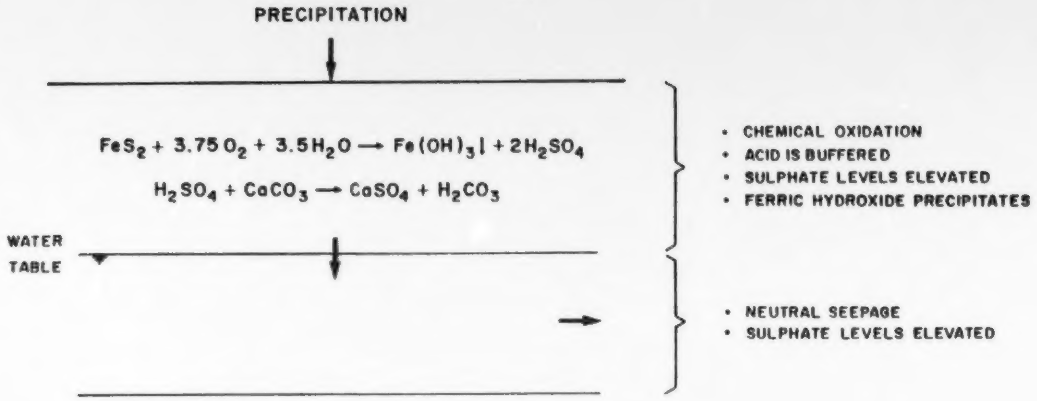
3.0 STAGES IN ACID GENERATION

Virtually all tailings and waste materials have some buffering capacity when placed into disposal areas. If placed under alkaline conditions (i.e. from pH 7 to 10) the pyritic material oxidizes chemically at a low but significant rate (Cherry et al., 1983; SENES, 1984). It is hypothesized that bacterial activity does not contribute significantly to the acid generation in this pH range unless elemental sulphur or thiosalts are present (Arkesteyn, 1980). Since chemical oxidation of ferrous ion is very rapid above pH 7 and ferric ion is almost instantaneously precipitated from solution as an hydroxide, the rate of pyrite oxidation is controlled at a low rate by the loss of the ferric ion. Runoff or seepage characteristics during this period are controlled by the buffering characteristics of the waste and the rate of flushing by precipitation. Typically runoff waters would contain elevated levels of sulphate and pH levels in the 5 - 8 range. Iron is present as Fe^{3+} and would be precipitated or reduced to Fe^{2+} in the saturated zone.

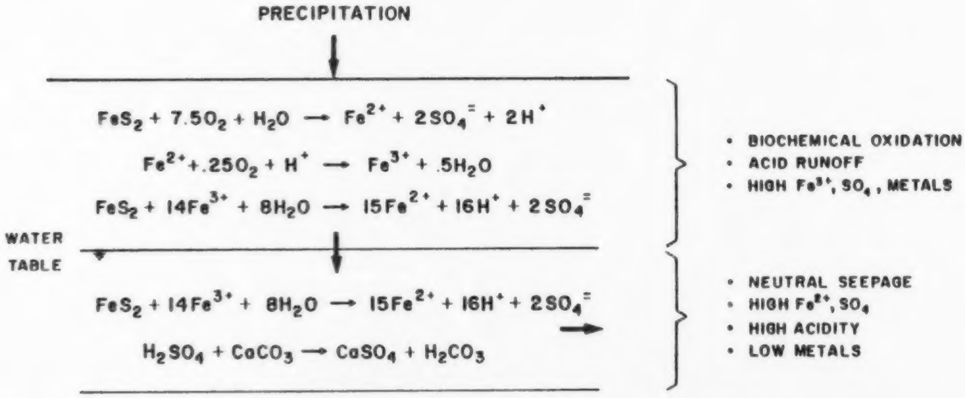
As the buffering capacity is depleted and the pH in the surface zone begins to drop, bacterial oxidation accelerates the rate of pyrite oxidation and greatly increases the rate of ferrous ion oxidation. The surface zone becomes acidic and pore waters may become highly contaminated. The pH levels are typically in the 1.5 to 4 range. Heavy metals are dissolved and iron is present as both ferric and ferrous ion with ferric ion often being the predominant species in surface runoff.

As seepage from this zone passes downward, the pH will increase as buffering occurs and iron may be precipitated as ferric hydroxide. As this acid front moves downward, ferric hydroxide is redissolved and passes into the saturated zone or zones where oxygen is depleted. Further buffering occurs in the saturated zone and ferric ion is reduced to ferrous ion with pyrite oxidation. As long as buffering is continuing, this pore water may be near neutral pH yet contain many thousands of mg/L of acidity, ferrous ion and sulphate. Heavy metals whose solubility is pH or sulphate dependent are typically at low concentrations. Seepages emerging into a oxidizing environment will rapidly oxidize resulting in a decline in pH levels to the acid range. Eventually as the receiving waters buffer the acidity, ferric hydroxides will precipitate leaving the characteristic gypsum and iron sludges so commonly associated with acid mine drainage.

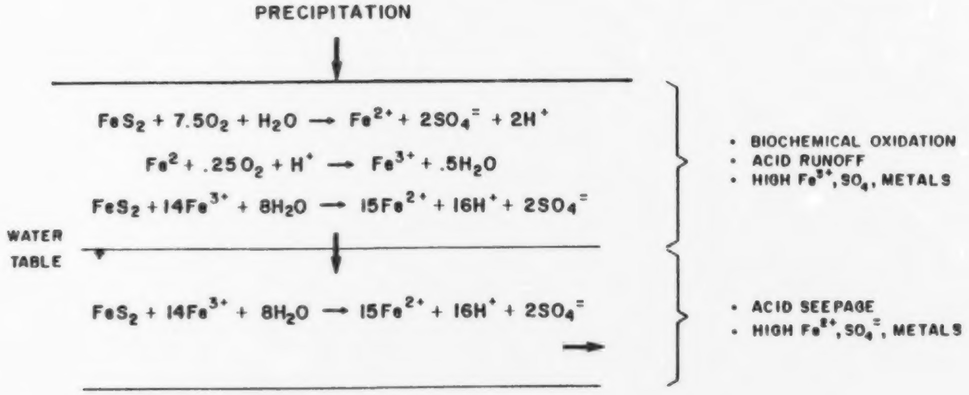
The progress of the acid generation as it relates to porewater quality as shown graphically on Figure 2.


PRECIPITATION
$FeS_2 + 3.75\,O_2 + 3.5H_2O \rightarrow Fe(OH)_3\downarrow + 2H_2SO_4$
$H_2SO_4 + CaCO_3 \rightarrow CaSO_4 + H_2CO_3$
WATER TABLE
• CHEMICAL OXIDATION
• ACID IS BUFFERED
• SULPHATE LEVELS ELEVATED
• FERRIC HYDROXIDE PRECIPITATES
• NEUTRAL SEEPAGE
• SULPHATE LEVELS ELEVATED

A) THE ONSET OF PYRITE OXIDATION


PRECIPITATION
$FeS_2 + 7.5O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{=} + 2H^+$
$Fe^{2+} + .25O_2 + H^+ \rightarrow Fe^{3+} + .5H_2O$
$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 16H^+ + 2SO_4^{=}$
WATER TABLE
$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 16H^+ + 2SO_4^{=}$
$H_2SO_4 + CaCO_3 \rightarrow CaSO_4 + H_2CO_3$
• BIOCHEMICAL OXIDATION
• ACID RUNOFF
• HIGH Fe^{3+}, SO_4, METALS
• NEUTRAL SEEPAGE
• HIGH Fe^{2+}, SO_4
• HIGH ACIDITY
• LOW METALS

B) ACID CONDITIONS OCCURRING


PRECIPITATION
$FeS_2 + 7.5O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{=} + 2H^+$
$Fe^2 + .25O_2 + H^+ \rightarrow Fe^{3+} + .5H_2O$
$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 16H^+ + 2SO_4^{=}$
WATER TABLE
$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 16H^+ + 2SO_4^{=}$
• BIOCHEMICAL OXIDATION
• ACID RUNOFF
• HIGH Fe^{3+}, SO_4, METALS
• ACID SEEPAGE
• HIGH Fe^{2+}, $SO_4^{=}$, METALS

C) BUFFERING DEPLETED

FIGURE 2

THE STAGES OF ACID GENERATION

4.0 BIOCHEMICAL FACTORS AFFECTING ACID GENERATION

Section 2 briefly outlines the factors which can affect the rate of pyrite oxidation by both biological and chemical methods. These various factors have been critically reviewed by SENES *et al.* (1986) for the National Uranium Tailings Program Office, in the development of the RATAP (Reactive Acid Tailings Assessment Program) acid generation model. The primary factors affecting acid generation rates are reviewed in this section.

i) Temperature

Investigators have reported the optimum temperature for biological ferrous ion oxidation by *T. ferrooxidans* to lie between 33 and 40°C (Malouf and Prater, 1961; Landesman *et al.*, 1966; MacDonald and Clark, 1970). Leaching studies on metal sulphide ores and concentrate samples have demonstrated that

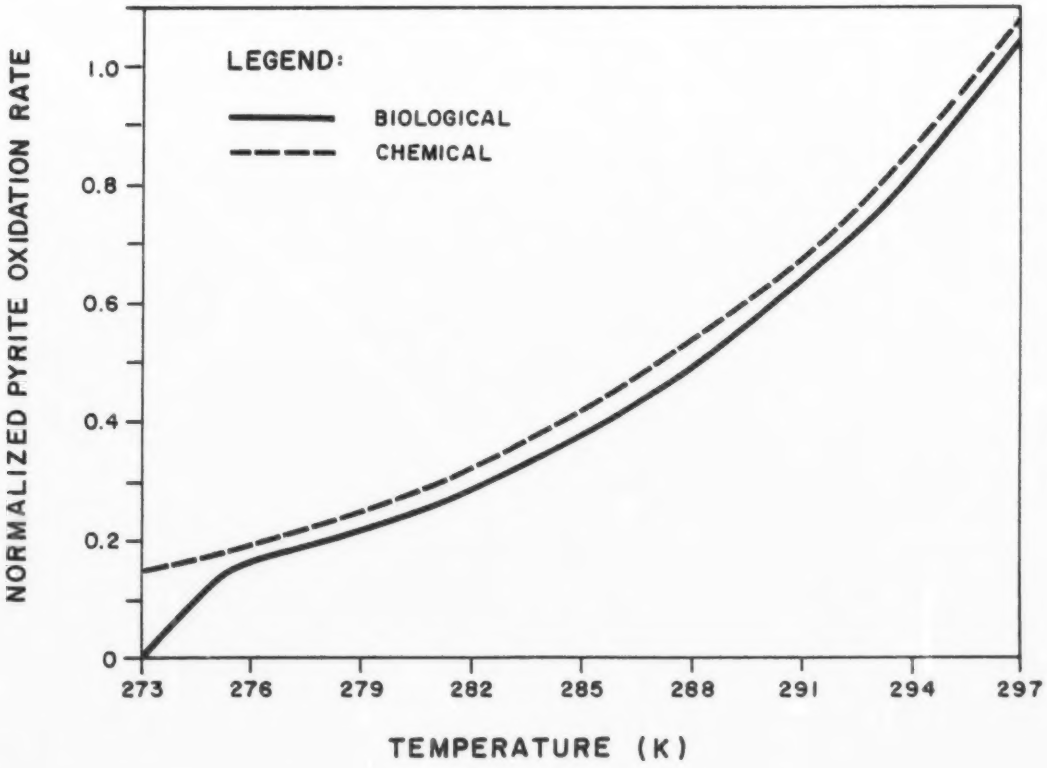


FIGURE 3

EFFECT OF TEMPERATURE ON BIOLOGICAL AND CHEMICAL OXIDATION RATE the optimum temperature generally is in the range of 30 to 35°C (Razzell and Trussell, 1963; Torma et al., 1970).

The influence of temperature on chemical oxidation is similar to that of bacterial oxidation; over the range of temperatures comparable to natural conditions, the rate approximately doubles for each 10°C rise (Morth and Smith, 1966; Konecik, 1965).

The effects of temperatures on biological and chemical oxidation rates are shown graphically on Figure 3.

ii) pH

T. ferrooxidans, the main bacteria catalyzing-biological pyrite oxidation has an optimum growth rate in the acidic pH range (i.e. pH 1.5 to 5.0) (SENES, 1984). In studies by Arkesteyn (1979) the rate of bacterial oxidation of pyrite by *Thiobacillus ferrooxidans* was observed to be a maximum over a pH range from 3.0 to 3.5. The work of Silverman and Lundgren (1959), Landesman et al. (1966), and Malouf and Prater (1961), suggest that the maximum activity falls in the pH range between pH 2.5 and 4.0. The particular range of pH over which the growth of *T. ferrooxidans* is optimized is a function of the type of substrate (SENES, 1984).

Jaynes et al. (1984) fitted a quadratic equation to the pH data given by Silverman and Lundgren (1959a) and Schnaitman et al. (1969) for the pH range $1.54<pH<4.95$. It is:

$$X_{pH} = -0.35\ pH^2 + 2.3\ pH - 2.7$$

where

X_{pH} = fraction of peak cell activity as influenced by pH (0 to 1)

For a modified form this equation SENES et al. (1986) found the maximum cell activity ($X_{pH} = 1.0$) occurs at pH = 3.25.

Below pH 3.0, changes in pH have little effect on the chemical oxidation rate (Smith, Svanks and Shumate, 1968). However, above pH 3.0 the oxidation rate increases, although slowly at first, to the point where the rate doubles for every two pH units above pH of 6.0 (U.S. EPA, 1971).

Smith and Shumate (1970) demonstrated that the rate of reaction for the chemical oxidation of pyrite in water at 25°C and 0.1 MPa oxygen partial pressure increases nonlinearly, as the pH is raised from 1 to 10.

The effects of temperature on the biological and chemical oxidation rate are shown graphically on Figure 4.

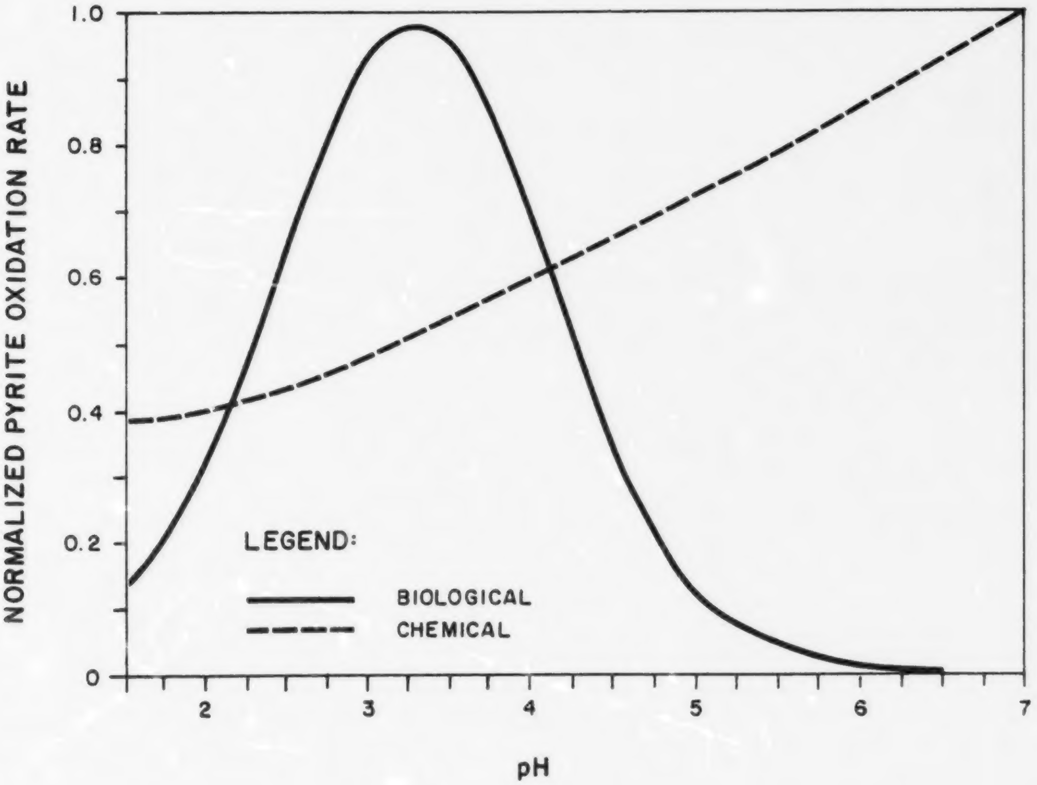


FIGURE 4
EFFECT OF pH
ON BIOLOGICAL AND CHEMICAL OXIDATION RATE

iii) Oxygen

The biological oxidation is not normally limited by oxygen concentration as the oxygen does not become a limiting substrate except at low levels. Myerson (1981) reports the limiting concentration to be 5% of the water saturation levels. Below this level the rate is a first order reaction with respect to oxygen concentration.

Chemical oxidation is directly proportional oxygen concentration where

$$\text{Rate} = K \cdot P_{O2}{}^{n}$$

where

K = rate constant

P_{O2} = partial pressure of oxygen

n = order of the reaction

Suggested values of n include .8, .67, and .5 (Lowson, 1982).

iv) Carbon Dioxide (Biological Effect)

Carbon dioxide is the main carbon source used by autotrophic thiobacilli for growth. The importance of carbon dioxide as a limiting nutrient is dependent on the geochemistry of the waste material. If the waste material contains a calcareous component, then an ample supply of CO_2 should be available (Harries and Ritchie, 1983).

Little information exists in the literature on the effect of very low fluxes of carbon dioxide on bacterial activity. Oxidation of pyritic material has been shown to proceed in the laboratory under conditions where there is a low carbon dioxide content in the gaseous phase. The rate of oxidation however, was much slower than observed with no restriction on carbon dioxide (Kelly and Jones, 1978).

Several researchers have found that bacterial activity can be increased by raising the carbon dioxide content in the gas phase. Torma et al. (1972) observed that the optimum carbon dioxide content in the gaseous phase, was about 0.22 percent, over the range of 0.03 to 7.92 percent. At CO_2 contents above 0.22 percent, there was no further increase in bacterial growth rate occurred. Approximately a two-fold increase in the leach rate was measured as the carbon dioxide content was increased from 0.03 to 0.22 percent.

v) Nutrients (Biological Effect)

Nutrient requirements (such as nitrogen and phosphorus) for *T. ferrooxidans* are usually not limiting as extremely small concentrations are quite sufficient for growth (Nordstrom, 1982). Tuovinen and Kelly (1973) observed that *T. ferrooxidans* grow in the absence of any added N compound and suggested that they may fix atmospheric nitrogen.

Hoffmann *et al.* (1981) carried out experiments to determine the optimal N/P ratio for maximum acceleration of the rate of leaching and minimum inhibition by phosphate complexes and precipitates. As the N/P molar ratio was increased initially, the leaching rate increased until a maximum was reached at an N/P ratio equal to approximately 90 to 1. Above this ratio, the reaction rate decreased to a steady value at molar ratios greater than approximately 150 to 1.

vi) Water Content

The moisture content can be an important variable in determining microbial activity in tailings. Most microorganisms require water because of their relatively permeable cell membranes and large surface-to-volume ratio (Scow, 1982). Conversely, moisture can limit the rate of oxygen transport through the soil pore space and thus reduce the level of microbial activity.

The measure of soil moisture that is most relevant to describing the effect of water content on microorganisms is not moisture content but "water potential". The limits of soil water potential on most bacteria are -8×10^6 Pa (-80 bar), as the lower limit, and approximately -5×10^5 Pa (-5 bar), as the upper limit (Gray, 1978).

In a study by Belly and Brock (1974) the activity of T. ferrooxidans in coal refuse materials was measured at different water potentials. The study concluded that T. ferrooxidans resemble many other common soil bacteria in requiring a fairly high water potential for their activity. A minimum water potential for growth of -2.3×10^6 Pa (-23 bars) was observed under conditions analogous to a coal refuse pile (Brock, 1975).

Based on the experiment data of Belly and Brock (1974), a linear relationship between water content and the rate of biological pyrite oxidation is predicted by the model. No data are available on the influence of water content on the chemical rate of pyrite oxidation.

vii) Inhibitors

Several organic and inorganic compounds are known to inhibit T. ferrooxidans activity. Among the most inhibitory compounds are organic and amino acids (Matin, 1978). At organic acid (e.g. acetate and butyrate) concentrations as low as 0.1 $mol.L^{-1}$, the growth of T. ferrooxidans can be inhibited (Matin, 1978).

In contrast to most other bacteria, the tolerance of T. ferrooxidans to toxic metal ions is remarkable. Various strains have been reported to tolerate 1 $mg.L^{-1}$ mercury, 50 to 100 $mg.L^{-1}$ selenium, tellurium, and arsenic anions (Brierley, 1978), 200 $mg.L^{-1}$ silver (Sugio et al., 1981a), 0.5 to 1.0 $g.l^{-1}$ uranyl and thorium ions (DiSpirito and Tuovinen, 1982), 2 $g.L^{-1}$ of molybdenum and 5 $g.L^{-1}$ of chromium (Lundgren and Silver, 1980), 10 to 30 $g.L^{-1}$ of aluminum, cobalt, copper, manganese, and nickel (Lundgren and Dean, 1979; Lundgren and Silver, 1980) and from 70 to 120 $g.L^{-1}$ of zinc in solution (Torma et al., 1970 and 1972).

Although metal ion tolerance varies with the particular strain and environmental conditions, T. ferrooxidans possess a well noted ability to adapt to increasingly higher metal levels with each successive generation.

Some evidence suggests that this adaptability is due to plasmid encoding proteins that render intracellular toxic metals inactive (Torma, 1983). Plasmid Deoxyribonucleic Acid (DNA) has been found in 11 of 15 strains of _T. ferrooxidans_ (Martin _et al_., 1981). While plasmid was not detected in the other four, it may have been present.

viii) Surface Area and Mass

It is well documented that the rate of pyrite oxidation is a function of pyrite surface area (Hoffmann _et al_., 1981; Steger and Desjardins, 1978; Cornelius and Woodcock, 1958; Mathews and Robins, 1974). As oxidation proceeds the surface area of the pyrite particles decreases. The rate at which the pyrite particle "shrinks" as it is depleted through oxidation can be described by the following expression:

$$\frac{dM}{dt} = \frac{\rho_p A_p}{V} \left(\frac{dr}{dt}\right)$$

where:

M = mass to pyrite per unit volume of tailings

ρ_p = mass of pyrite per unit volume of pyrite

r = average radius of pyrite particles

V = volume of tailings

A_p = total surface area of pyrite particles

In the above expression the pyrite particles are normally assumed to be spherical. Because the small grains with a proportionally higher surface area to mass ratio of pyrite are depleted quickly, large reductions in surface area can be achieved with relatively small changes in mass. Therefore it is essential to know the distribution of pyrite grain sizes so that a relationship between available surface area, mass, and rate can be developed.

5.0 CONCLUSION

The geochemistry of acid generation in tailings and waste rock is reasonably well understood. Recent studies have greatly improved the level of knowledge relating to factors affecting acid generation rates. With this knowledge, new models are being developed which will greatly assist the mining industry in both the assessment of the extent of acid generation potential and development of control technologies.

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ACID MINE DRAINAGE IN BRITISH COLUMBIA

TODAY AND TOMORROW

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ABSTRACT

Of the 16 metal mines currently operating in British Columbia, five generate acid mine drainage. At least five abandoned mines also produce acid mine drainage. Acid mine drainage flows from underground workings, tailings, open pits and waste rock dumps. The production of acid mine drainage from open pits and waste rock dumps is a major concern in B.C.

The date of initial release of acid mine drainage is uncertain for most mines, but is known to be as rapid as one year and as slow as 11 years. Significant levels of *Thiobacillus ferrooxidans* have been found in all acid mine drainage.

Release of acid mine drainage has resulted in some major impacts on the environment in B.C. The five operating mines have instituted comprehensive pollution abatement programs to address the problem while little has been done to control the abandoned mines.

New mine proposals are now reviewed in detail. Many of the properties currently under active exploration have the potential for acid generation. Potential acid generating mines are primarily massive sulphide deposits although there are three possible coal mines with acid mine drainage potential.

ACID MINE DRAINAGE IN BRITISH COLUMBIA - TODAY

Mining is a major resource industry in British Columbia. There are currently 23 mines operating in British Columbia. Of the 16 metal mines, five, or about 30 percent, generate acid mine drainage. In addition, at least five abandoned metal mines produce acidic drainage. None of the seven operating coal mines produce acid mine drainage as the coal contains relatively low levels of sulphur (mean 0.37 percent sulphur).

The acid generating metal mines are distributed throughout the province (Figure 1). Sources of acid mine drainage include underground workings, tailings, open pits, and waste rock dumps. The number of cases associated with open pits and waste rock is unique to British Columbia as the remainder of Canadian provinces have few large open pit operations and associated waste rock dumps. In B.C., the total quantity of acid generating tailings is about 72,000,000 tonnes (Table 1) and waste rock is 229,000,000 tonnes (Table 2). These figures do not consider other sites suspected by the authors but unconfirmed. Acid generating tailings generally contain higher concentrations of sulphur than waste rock. Two acid generating waste rock dumps apparently contain less than two percent sulphur. The porous nature of waste rock

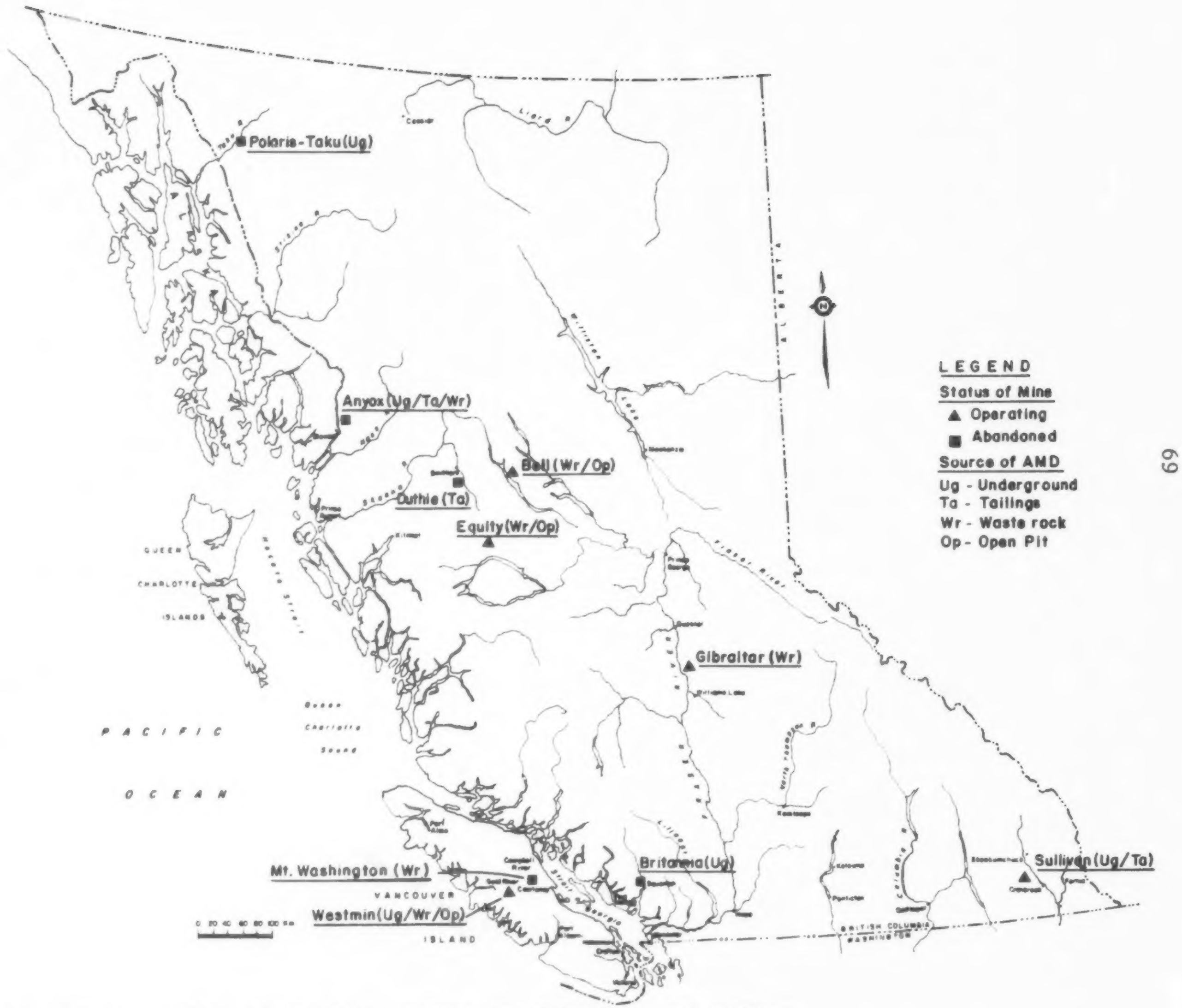

Polaris-Taku(Ug)
Anyox(Ug/Ta/Wr)
Bell(Wr/Op)
Duthie(Ta)
Equity(Wr/Op)
Gibraltar (Wr)
Britannia(Ug)
Mt. Washington (Wr)
Westmin(Ug/Wr/Op)
Sullivan(Ug/Ta)
LEGEND
Status of Mine
▲ Operating
■ Abandoned
Source of AMD
Ug - Underground
Ta - Tailings
Wr - Waste rock
Op - Open Pit
ALBERTA
N
QUEEN CHARLOTTE ISLANDS
Hecate Strait
Queen Charlotte Sound
PACIFIC OCEAN
VANCOUVER ISLAND
Fraser River
BRITISH COLUMBIA
WASHINGTON
0 20 40 60 80 100 Km

FIGURE I ACID PRODUCING MINES IN BRITISH COLUMBIA

TABLE 1

ACID GENERATING TAILINGS IN BRITISH COLUMBIA

Mine	%S	Quantity (tonnes)
Sullivan	28.0%	60,000,000
Anyox*	UNK	12,000,000
Duthie	8.0%	80,000
	TOTAL	72,080,000

* most of the Anyox tailings were deposited under water

TABLE 2

ACID GENERATING WASTE ROCK IN BRITISH COLUMBIA

Mine	%S	Quantity (tonnes)
Gibraltar	1.5%	91,000,000
Equity Silver	4.0%	70,000,000
Noranda - Bell	UNK	60,000,000
Westmin	5.0%	8,000,000
Mt. Washington	1.2%	400,000
	TOTAL	229,400,000

dumps allows oxygen penetration to much greater depths than in tailings. As such, waste rock dumps may generate as much acid mine drainage per unit surface area as tailings though sulphur levels are much lower.

The date of initial release of acid mine drainage is unknown for most mines. However, lag from mine start-up to the discovery of acid mine drainage is known to be about 11 years for the Gibraltar mine and only about 1 year for the Equity mine (Table 3). All mines have been generating acidic drainage for several years and four have done so for more than a decade. At the Britannia mine, the acidic minewater was reported to contain from 20 to 170 mg/l dissolved copper in 1955, similar to levels found in 1966 (Figure 2). Since 1966, levels have generally decreased although there is evidence of a levelling off since 1979. A smooth curve through the data suggests elevated levels of copper will continue to be released for several more decades.

Virtually all acid mine drainage produced exceeds federal and provincial effluent discharge criteria by several orders of magnitude (Table 4). The quality is variable reflecting differences in the mineralogy of the mining wastes. As a general rule, mines that are, or were, copper mines have

TABLE 3

LAG TO FORMATION AND DURATION OF SOME ACID MINE DRAINAGE IN BRITISH COLUMBIA

Mine	Acid Mine Drainage	Mine Start Up	Mine Abandonment	Acid Mine Drainage Generated	Lag To Formation (years)	Duration of Generation (years)
Britannia	minewater	1905	1974	<1928	<23	>58
Duthie	tailings seepage	1952	1954	1980	<26	> 6
Westmin	waste rock runoff	1966	operating	<1979	<13	> 7
Equity	waste rock runoff	1980	operating	1981	1	5
Gibraltar	waste rock runoff	1971	operating	1982	11	4
Mt. Washington	waste rock runoff	1964	1966	<1972	<6	>14
Sullivan	minewater	1909	operating	<1950	<41	>36
Anyox	minewater	1914	1936	<1972	<36	>14

TABLE 4

EXAMPLES OF SOME ACID MINE DRAINAGE IN BRITISH COLUMBIA

(Adapted from Ferguson and Mehling, 1986)

OPERATING MINES

MINE	LOCATION	SOURCE AND PEAK CONTAMINANT CONCENTRATIONS*			
		Underground Mine	Open Pit	Tailings	Waste Rock
Gibraltar	McLeese Lake				pH-3.1 acid-2675 Cu-257 Al-183 Fe-71
Equity Silver	Houston		pH-3.0 acid-364 Cu-17.5 Zn-9.9		pH-2.4 acid-44100 Cu-791 Zn-350
Westmin	Myra Falls	pH-6.9 acid-61 Zn-37 Cu-4 Al-5 Fe-16	pH-3.0 acid-423 Zn-51 Cu-10 Al-20 Fe-72		pH-3.8 acid-397 Zn-74 Cu-13 Al-36 Fe-9.5
Noranda (Bell)	Granisle		pH-6.9 acid-26 Cu-0.8		pH-3.3 acid-185 Cu-10.6 Fe-36.7
Sullivan	Kimberley	pH-3.1 acid-4280 Zn-69 Fe-191 Mn-25 Al-88 Pb-6.4		pH-2.7 acid-995 Zn-76 Fe-817 Mn-44 Al-71 Pb-5.3	

CONTINUED . . . .

TABLE 4

EXAMPLES OF SOME ACID MINE DRAINAGE IN BRITISH COLUMBIA

(Adapted from Ferguson and Mehling, 1986)

ABANDONED MINES

MINE	LOCATION	SOURCE AND PEAK CONTAMINANT CONCENTRATIONS*			
		Underground Mine	Open Pit	Tailings	Waste Rock
Britannia	Britannia Beach	pH-2.7 acid-494 Cu-43 Zn-32 Fe-45			
Duthie	Smithers			pH-1.9 acid-N.A. Cu-138 As-5060 Zn-1360 Fe-13300	
Anyox	Anyox	pH-N.A. acid-N.A. Cu-1.7 Zn-3.4 Al-37 Fe-195		pH-N.A. acid-N.A. Cu-3.1 Zn-5.9 Al-150 Fe-1650	pH-N.A. acid-N.A. Cu-2.17 Zn-3.74 Al-45.6
Mt. Washington	Courtenay				pH-3.4 acid-431 Cu-77 Al-50 Fe-7.2
Tulsequah	Taku River	pH-5.7 acid-43 Cu-2.97 Zn-14.1 Fe-0.4			

* Data from EPS surveys except Britannia (B.C. Waste Management Branch). Peak concentrations are undoubtedly higher than those found during infrequent EPS surveys. Units are mg/l for metals, acidity is mg/l as $CaCO_3$.

acid mine drainage with particularly elevated copper values while lead/zinc mines produce drainage high in those metals. Iron values are surprisingly low in the acidic drainage from some mines. Elevated concentrations of iron would be expected since the oxidation of pyrite is believed to play the critical role in acid mine drainage formation. At mines with low iron, ferric iron may precipitate in the waste or galvanic reactions between sulphide minerals may hinder pyrite oxidation. Eventually iron could be released from these wastes as the geochemical process matures.

The strongest acid mine drainage is produced from a waste rock dump at the Equity Silver mine. Very low pH, high acidity and metal values are found in the acid mine drainage through most of the year. Elevated internal temperatures and steaming vents indicate the great activity of the biologically catalyzed process at that mine.

Significant levels of the bacteria, Thiobacillus ferrooxidans, have been found in all acid mine drainage tested (Table 5). By contrast, neutral drainage contains low levels. These data illustrate the important role of bacteria in producing acid mine drainage. The iron oxidizing Thiobacillus ferrooxidans is believed responsible for acid generation at most mines. Tests for the sulphur oxidizing Thiobacillus thiooxidans and thermophilic sulfolobus

TABLE 5

THIOBACILLUS FERROOXIDANS IN DRAINAGE FROM SOME MINES IN BRITISH COLUMBIA

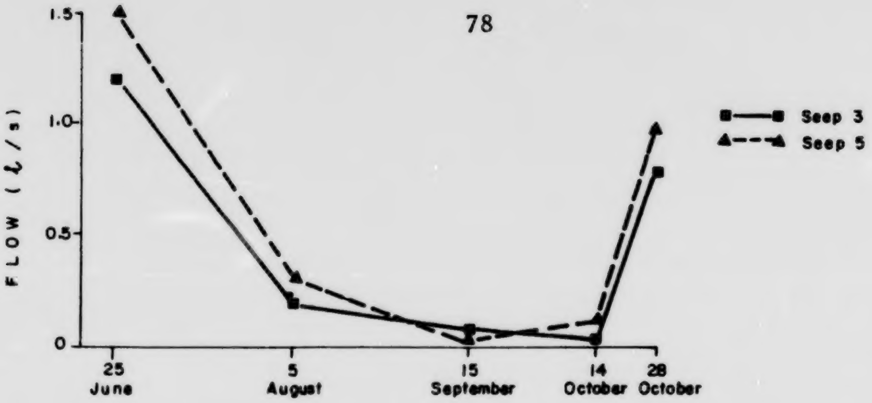
Mine	pH of Drainage	Thiobacillus ferrooxidans in Drainage*
Acidic Drainage		
Gibraltar	2.0 - 4.0	9,200 - 540,000
Noranda (Bell)	2.8 - 4.0	160,000 - 3,500,000
Mt. Washington	3.5 - 4.0	790 - 540,000
Sullivan	2.7 - 3.1	2,400,000 - ≥2,400,000
Equity	2.4 - 2.8	130,000 - ≥24,000,000
Neutral Drainage		
Noranda (Bell)	7.3	230
Sullivan	7.0	< 20
Equity	5.1	<200
Premier Silbak	6.2	<200

* MPN/100 ml

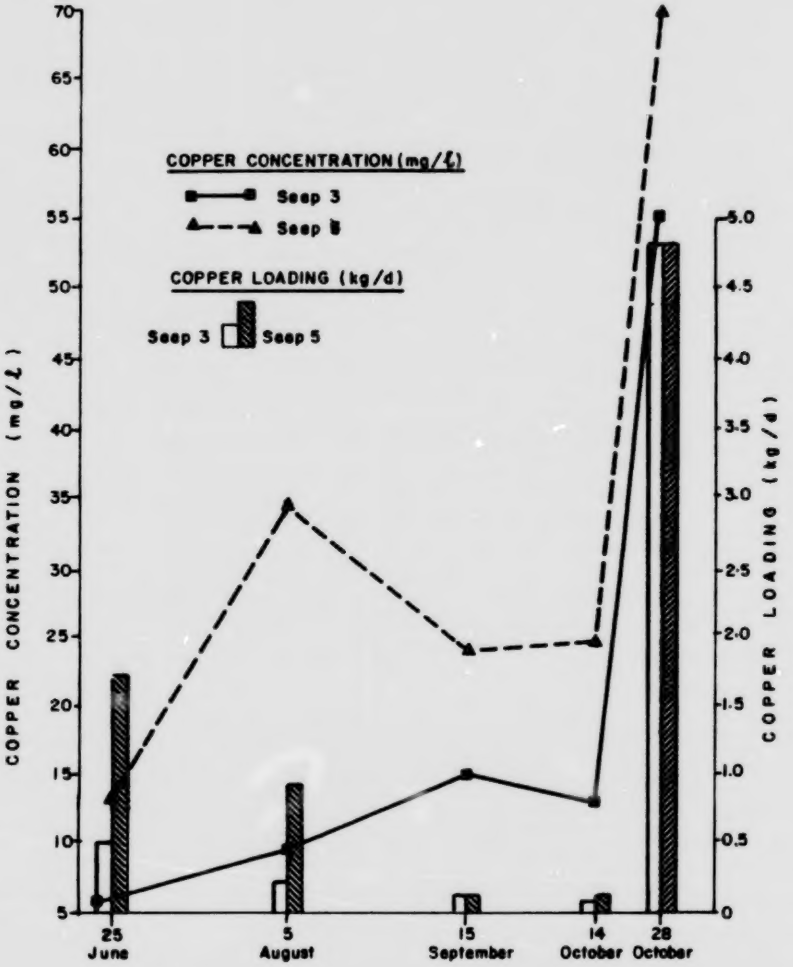
bacteria in acidic drainage from the Gibraltar mine indicated insignificant levels of those organisms (Kay, 1986).

Seasonal variations in contaminant concentration and loading have been observed at all mines. Seeps from the waste rock dump at the Mount Washington mine exhibit elevated copper concentration and load in the spring and fall associated with freshet and rainfall events respectively (Figure 3). The long relatively dry summer period allows the production and accumulation of soluble metal sulphate salts which are flushed from the dump during the first major precipitation event. In 1986, this waste rock dump produced about 20 tonnes of sulphuric acid from only 250,000 tonnes of waste rock based on the sulphate concentrations in the drainage from the dump.

The release of acid mine drainage has resulted in some major impacts on the environment in British Columbia. A review by Clark and Morrison (1982) of receiving water quality data collected near the Westmin mine indicated that zinc and copper levels in Buttle Lake and downstream waters had significantly increased. Both of the metals were in concentrations which posed an immediate threat to aquatic life in the watershed. A reduction in fishing success, and a general decrease and change in plankton community structure was also noted. Subsequent investigations found acid mine drainage



TEMPORAL VARIATION IN SEEPS 3 AND 5 FLOW



TEMPORAL VARIATION IN SEEPS 3 AND 5 COPPER LEVELS

FIGURE 3 MOUNT WASHINGTON WASTE ROCK DUMP SEEP FLOW AND COPPER LEVELS

from waste rock dumps and an open pit at the mine to be the primary sources of metals to the lake. Receiving water quality dramatically improved following the institution of a comprehensive acid mine drainage abatement program (B. Kelso and R. Van Dyk, personal communication).

At the abandoned Mt. Washington mine, acid mine drainage is affecting water quality in the Tsolum River, some 11 kilometres downstream. Total copper concentrations in the river are at times acutely toxic to salmonids, suggesting that poor fish returns to a local hatchery, and a declining steelhead fishery, may be partly due to acid mine drainage (Kangasniemi and Erickson, 1986).

The five operating mines in British Columbia have instituted comprehensive pollution abatement programs to address the problem while little has been done to reduce acid mine drainage from the abandoned mines. The five abandoned mines are severely impacting about 30 kilometres of streams. In addition, the marine environment adjacent to the Britannia and Anyox mines is impacted up to 18 kilometres and 6 kilometres from the minesites as indicated by elevated metal levels in molluscan bivalues (mussels, oysters, etc.) (D. Goyette, personal communication).

High precipitation in some parts of the province result in the formation of large volumes of acid mine drainage. Many British Columbia streams support valuable fisheries resources and are extremely sensitive to acid and metal pollution as they have low buffering and metals complexing capacities.

None of the operating acid-producing mines have reached a maintenance free state where they can be safely abandoned. At least 150 kilometres downstream of the Equity mine could be severely impacted by acid mine drainage if collection and treatment were to be suspended based on current acid production rates (F. Rhebergen and B. Wilkes, personal communication). Concern over long-term responsibility and potential environmental impacts has lead to more stringent review of new mining proposals, and may present serious impediments to the future development of British Columbia's sulphide mineral resources.

ACID MINE DRAINAGE IN BRITISH COLUMBIA - TOMORROW

Potentially acid generating wastes are currently being produced in British Columbia although no evidence of acidic drainage has yet been found. Tailings deposited at the Westmin mine have very high levels of sulphur and show segregation of sulphide and carbonate minerals based on particle size and density (Figure 4). These tailings are deposited by the subaerial method where thin layers of tailings are discharged from a spray bar system and allowed to drain and consolidate. Westmin hopes to produce a dense unsaturated tailings mass with low horizontal and vertical permeability. These physical parameters may limit acid mine drainage production. No acid mine drainage has been found from the tailings after two years of operation of the subaerial system. Further studies are required to ensure design criteria are met and these conditions prevent acid formation.

Tailings from the Equity, Scottie, and Island copper mines in British Columbia are also reported to be acid generating, but no acid mine drainage has been found to date. The former mine has maintained the alkalinity of the tailings pond supernatant and flooded most of the tailings during its operation. The latter two mines have placed tailings underwater. Although unconfined underwater disposal of tailings

SAMPLE	% S	NP*
Tailings Discharge at Spray Bar	10.5	45.7
Tailings Sands	32.0	16.3
Beach under Spray Bar	32.2	16.9

AT TAILINGS AREA No. 2

* Neutralization Potential Units are Tons $CaCO_3$ equivalent / 1000 Tons Material

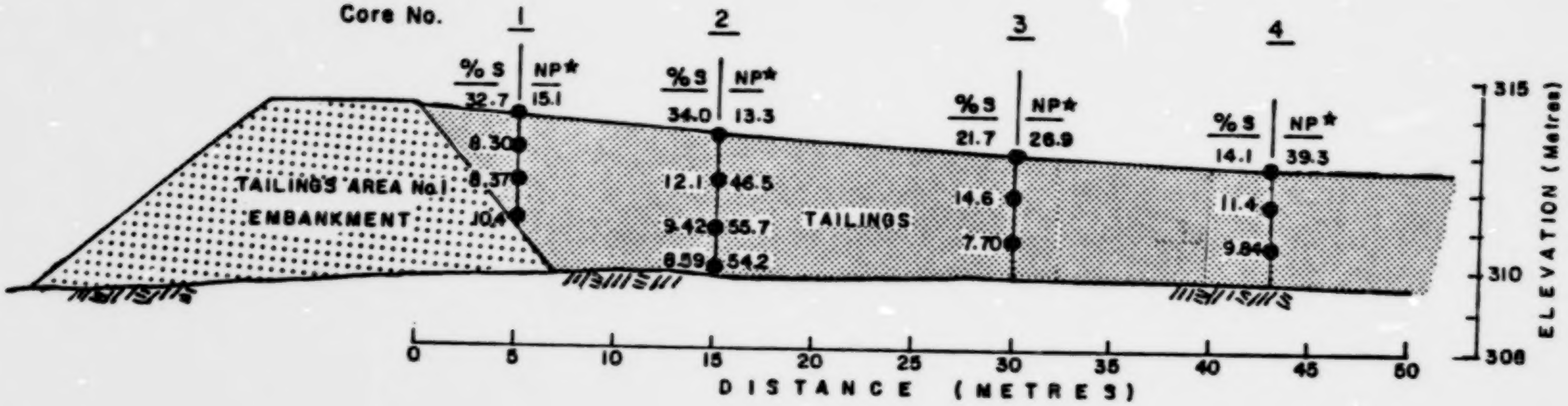

Core No.
1
% S
NP*
32.7
15.1
8.30
8.37
10.4
2
%S
NP*
34.0
13.3
12.1
46.5
9.42
55.7
8.59
54.2
3
%S
NP*
21.7
28.9
14.6
7.70
4
%S
NP*
14.1
39.3
11.4
9.84
TAILINGS AREA No. 1
EMBANKMENT
TAILINGS
0
5
10
15
20
25
30
35
40
45
50
DISTANCE (METRES)
315
310
308
ELEVATION (Metres)

FIGURE 4 ACID GENERATION POTENTIAL OF WESTMIN TAILINGS SAMPLES (Ferguson, 1986)

is a controversial issue in British Columbia, involving several potential impacts on the environment, the practice appears to be successful in preventing acid mine drainage generation.

The recent history of Acid Mine Drainage in British Columbia has indicated, if nothing else, the importance of assessing each individual mine on its own merits. There are certain criteria based upon the type of mineral deposit which provide a general indication of whether or not acid mine drainage could occur.

Massive sulphide deposits

These deposits are one of the major sources of acid mine drainage in British Columbia. Three of the five operating and four of the abandoned mines are massive sulphide deposits.

Operating Mines	Abandoned Mines	Potential future mines
Sullivan Westmin (Equity)	Britannia Anyox Tulsequah Mt. Washington	Kutcho Creek Cirque Lara Windy Craggy Rea Gold Chu Chua

The authors know of no massive sulphide mines which have not become acid producers. There are a number of properties which have been discovered and are undergoing feasibility studies at present, all of which are a major concern.

Porphyry Copper

There are nine large open pit porphyry copper mines in British Columbia. Two of these, Bell Copper and Gibralter are acid producers and at least one of the others has the potential for producing acid mine drainage. Equity Silver can be considered a transition between a shallow porphyry and a massive sulphide deposit. The future of large low-grade copper producers is not bright in British Columbia and no future porphyry copper mines are anticipated for the next decade.

The existing porphyry copper producers should continue to be monitored closely.

Vein Deposits

The Duthie property is an example of a mesothermal vein system and the authors could find no other good example of vein system which has the potential for acid mine drainage.

Skarn Deposits

Both Craigmont and Giant Mascot are example of skarn deposits and neither operation has shown any signs of acid generation. While pyrite levels are high, the high carbonate associated with skarn replacement deposits has undoubtedly neutralized any acid generation.

Epithermal Gold Deposits

With the current high price of gold, exploration for epithermal gold deposits has accelerated in B.C. Generally, epithermal deposits are relatively low in sulphide, however, as the depositional environment becomes lower, the concentration of sulphide increases. The Cinola property on the Queen Charlotte Islands is an example of an ephithermal type where acid generation is definitely a possibility. Epithermal deposits may be associated with carbonate rich or poor host rock.

Coal Deposits

As previously discussed, the seven operating coal mines in the southeast and northeast regions of British Columbia do not generate acid mine drainage. However, coal mines proposed in north central B.C. and on Vancouver Island have encountered sulphur levels from 0.6% to 4.2%.

MINE	SULPHUR LEVEL IN RAW COAL	
	Weighted Mean	Range
operating mines*	0.37%	0.20 - 0.59%
proposed mines**		
- Quinsam	1.40%	0.6 - 3.20%
- Telkwa	1.50%	0.7 - 4.20%
- Nuspar	--	0.6 - 2.30%

* from Paine and Blakeman, 1983
** from Environmental Impact Statements from respective mines

Concerns for possible acid generation from the Quinsam and Telkwa mines resulted in the preparation of comprehensive acid mine drainage prevention plans (Sturm, Environmental Services 1983 and 1985).

The proposed mines have not proceeded due to depressed world wide coal markets. Monitoring of drainage from a reclaimed bulk sample backfill at the Quinsam mine, however, found low levels of contaminants and suggests the acid mine drainage prevention plan could be successful for the full scale operation.

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Acid Mine Drainage In U.S. Coal Regions

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Acid mine drainage is a common problem in the U.S., particularly in the Appalachian region where both the coal and overlying strata tend to be relatively high in pyrite and low in carbonate mineral content. Of the three primary sources of stream degradation by acid mine drainage, underground mines, coal refuse disposal sites, and surface mines, the latter source has been died most intensively.

Two case studies are presented to illustrate contaminant formation and discharge in surface mine settings. The sites are typical small Appalachian mines partially reclaimed at closure. Case I represents a revegetated mine with drainage water quality dominated by pyrite oxidation localized in a small fraction of the backfill. Case II represents acid production disseminated across the mined area, to the extent that vegetation could not be sustained by conventional reclamation practices. Results of various monitoring methods are described to show the importance of water quality, spoil, and pore gas analyses in developing an understanding of contaminant formation and discharge processes.

Introduction

Acidic, metal-laden drainage is a common result of mining in the Appalachian coal field of the Eastern U.S. The source of acid mine drainage (AMD) is the mineral pyrite (FeS_2), present within the coal and overburden strata, which undergoes oxidation when exposed to oxygen and water. Acid formation generally begins during the mining process and continues long after production ceases. Surface discharges exhibit a wide range of water quality, from nearly the same as ambient stream quality to grossly contaminated.

Adverse effects of coal mining on water quality in the U.S. have been limited since 1977, following passage of the Surface Mining Control and Reclamation Act. Prior to this Federal act or similar State legislation, over 5,000 miles of streams in the U.S. had been contaminated by mine drainage (Appalachian

Regional Commission, 1969; Kim et al, 1982). Of the pre-law sources, underground mines constitute the largest contributor to stream quality degradation, followed by preparation plant waste disposal sites, and surface mines.

Drainage from underground mines tends to be high in volume and moderate in contaminant concentrations. The relative inaccessibility of the underground environment hampers detailed understanding of acid formation and discharge processes, particularly after mine closure. Disposal sites comprised of coal cleaning wastes often produce the most concentrated acid drainage. We have observed sulfate and acidity concentrations in the range of 1,000 to 20,000 mg/L at many coal refuse dumps. Waste dumps often provide ideal weathering conditions: high surface exposure to oxygen and water, small particle sizes, and potential for physical weathering to expose new reaction surfaces.

Drainage quality from surface coal mines in potentially acid-producing regions ranges from almost no degradation to acidity and sulfate levels in excess of 1,000 mg/L. As yet, there is no clear understanding of why similar materials can produce radically different water quality effects. The nature of the materials, their juxtapositioning in the backfill, and the weathering potential of the post-mining environment all contribute to determining water quality. The following two case studies illustrate acid mine drainage formation and discharge processes in typical surface mine settings.

Case Study I: Upshur County, West Virginia

This site is a regraded, revegetated surface mine operated in the late 1970's. The Lower and Middle Kittanning seams were mined from the U-shaped, 6-ha site.

Post-mining topography consists of a 12-m slope at the highwall, a bench dipping back gently toward the highwall, and a 12-m outslope leading to toe-of-spoil seeps (Fig. 1). Spoil thickness in the bench area ranges from 4.3 to 7 m. Hydrologic features include recharge by precipitation and ground water flow; impoundment of water on the intact fireclay formerly underlying the Lower Kittanning coal; and discharge at the toe of the spoil in several well-defined seeps.

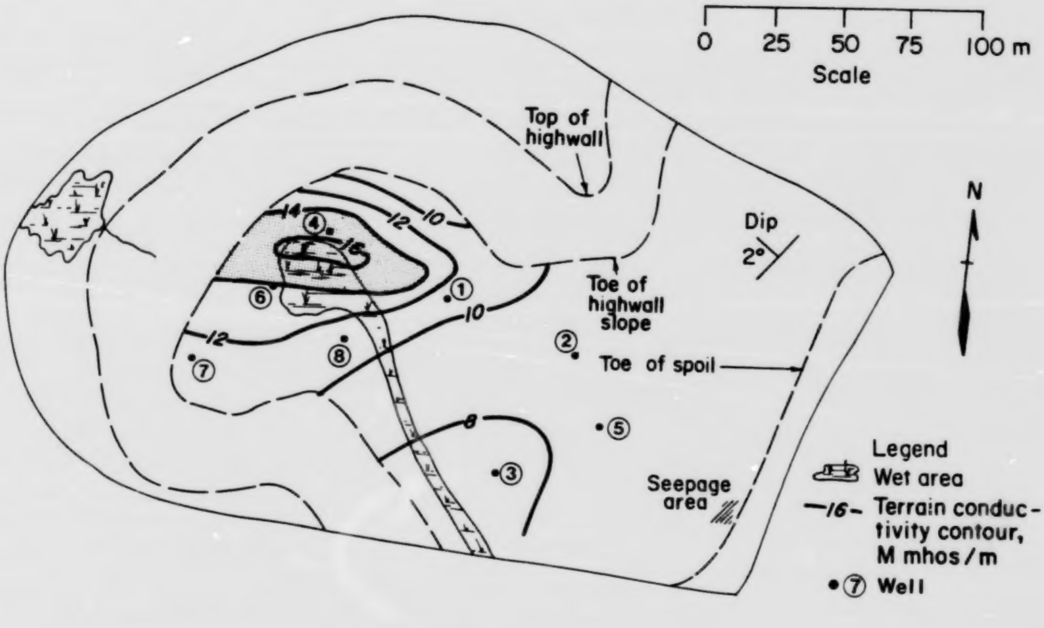
0
25
50
75
100 m
Scale
Top of highwall
Toe of highwall slope
Dip
2°
N
Toe of spoil
Seepage area
14
12
10
16
10
12
8
①
②
③
④
⑤
⑥
⑦
⑧
Legend
Wet area
-16- Terrain conductivity contour, M mhos/m
•⑦ Well

Figure 1. Map of the Upshur County site. High terrain conductivity, indicated by shading, is suggestive of acid-producing material (Ladwig, 1982; Ladwig, 1985).

A detailed site investigation was conducted over a 2-year period. This included application of surface geophysics for acid source location (Fig. 1; discussed elsewhere in these Proceedings), spoil composition analysis, pore gas analysis, and hydrologic monitoring within the backfill and at the discharge points. Monitoring well locations are shown in Figure 1.

Water quality data are summarized in Table 1. The two principal seeps, designated S1 and S1A, each discharged at an average of 0.98 L/s (15.5 gpm). Although the pH was quite low, contaminant levels were moderate in terms of the range observed for all mining wastes. The acidity-to-sulfate ratio, initially 1.04 when only pyrite reacts in unbuffered water, showed that slightly over half of the acidity produced with the sulfate had been neutralized within the backfill. Manganese and aluminum concentrations were also elevated, indicating acid-induced dissolution of other minerals.

Table 1. Mean water quality for the Upshur County site for the period November 1982 through April 1985.*

Sampling station	pH	Acidity	Iron, total	Sulfate	Aluminum	Manganese
Seeps						
S1	3.2	190.	26.	420.	11.	12.
S1A	3.0	250.	30.	550.	14.	18.
Wells						
1	6.3	55.	96.	400.	1.0	15.
2	4.2	260.	11.	480.	34.	16.
4	3.5	630.	252.	1100.	20.	15.
6	5.4	130.	67.	310.	<1.	10.
7	5.9	10.	19.	92.	<1.	5.0
8	6.0	71.	53.	250.	<1.	11.

*All concentrations are in mg/L rounded to 2 significant figures; pH is in standard units; acidity is reported as calcium carbonate equivalent.

Water quality was much more variable in monitoring wells than in the seeps. Mean sulfate concentrations ranged from 92 to 1100 mg/L, with the low extreme representing spoil water near the ground water recharge point. A high value, such as that observed at well 4, can usually be interpreted to be a pyritic zone actively undergoing oxidation.

Analyses of drill cuttings, taken during monitoring well installation, by the acid-base account method of overburden analysis (Sobek et al 1978) showed most of the spoil to be relatively low in pyrite, less than 0.3 pct by weight total sulfur. Neutralization potentials were also low, with the highest value being 0.45 pct calcium carbonate equivalent. In contrast to the bulk spoil, black material buried in the location of wells 4 and 6, and the Lower Kittanning underclay were found to have sulfur contents as high as 1.4, 4.0, and 2.8 pct, respectively. We suspect that the black material consists of pavement scrapings buried selectively for unknown reasons. This potentially acid-producing material is concentrated in the spoil at elevations within 3 meters of the surface and is intercepted by the saturated zone during high flow conditions.

Gas composition within the spoil pore spaces, measured at the bench area (between wells 8 and 4, fig. 1) and the toe outslope (well 5, fig. 1), showed distinctly different oxygen profiles. The oxygen content of gas in spoil on the bench decreased with increasing depth, reaching an average of less than 2 pct by volume within 1.5 m below surface. In contrast, the outslope spoil was oxygenated to more than 10 pct by volume throughout its unsaturated thickness. The sampling station on the bench was close to the acid-producing zone identified by spoil analyses and water quality measurements. The steepness of the oxygen gradient with depth is therefore considered to result from rapid consumption of oxygen by pyrite oxidation in the near-surface spoil. The more highly oxygenated state of the outslope spoil probably results from two factors: lower rates of oxygen consumption by pyrite (sulfur > 0.3 pct in this area) and higher rates of oxygen diffusion through the steep slope face. Had highly pyritic material been placed in this zone, the oxygenated environment would have likely supported a sharp increase in contaminant levels in the seeps relative to the nearest wells.

We attempted to control contaminant formation at this site by use of a bacterial inhibitor applied directly over the acid producing zone (Erickson and Ladwig, 1985). The results showed no subsequent improvement in water quality and, in fact, no migration of the inhibitor into the flow system. We concluded that the clayey soil cover adsorbed the active agent, precluding its migration to the oxidation zone. Similarly disappointing results might be expected with surface application of other ionic chemical treatments.

Other potential control strategies include excavation or in-place sequestering of the relatively small amount of spoil having a high acid potential. Excavation would still leave the problem of disposal of the pyritic waste and the possibility of continued low rates of acid production from the remaining spoil. Use of a barrier, such as an injectable grout, to isolate the source material from oxygen and water is an attractive but unproven control method.

The best means of improving the discharge quality at this site may be to let nature do the work. Our first sampling at the site in 1980 showed sulfate concentrations about double the average values found during the course of the study conducted between 1982 and 1985. Monitoring over a 10-year period would be needed to confirm a real trend in decreased contaminant formation. Construction of a wetland-based treatment system (described elsewhere in these Proceedings) would safeguard local water resources as natural abatement processes are monitored.

Case Study II: Clarion County, Pennsylvania

This site is a 12-ha hilltop mine that was operated in the mid-1960's. The Upper Clarion seam was mined across the entire site; the Lower Clarion seam was extracted from about one-third of the area (fig. 2). On completion, the backfill was graded to approximate original contour and seeded. The vegetation died out quickly and the site remained barren until 1984. In 1982 we undertook, in cooperation with the Pennsylvania Department of Environmental Resources, a study of acid production at the site as baseline monitoring prior to reclamation by the State (Lusardi and Erickson, 1985). Only pre-reclamation data are discussed below.

The site is a steeply-sided hill with a maximum spoil thickness of about 10 m near the center. Recharge is solely by infiltration and a shallow saturated zone is perched on the mine floor; less than 1 m of the Upper Clarion and less than 3 m of the Lower Clarion spoil are saturated. The spoil water discharges at a number of perimeter seeps, the largest of which are shown on Figure 2. The monitoring program included measurement of water quality, oxygenation of the spoil, and spoil composition.

Like the Upshur County site, this mine showed lateral variability in water quality, with the highest contaminant levels observed in monitoring wells and moderate contaminant levels at the discharge points (Table 2). In addition, water quality varied over time to the extent that the maximum sulfate concentration was two to 12 times higher than the minimum concentration at the same sampling station.

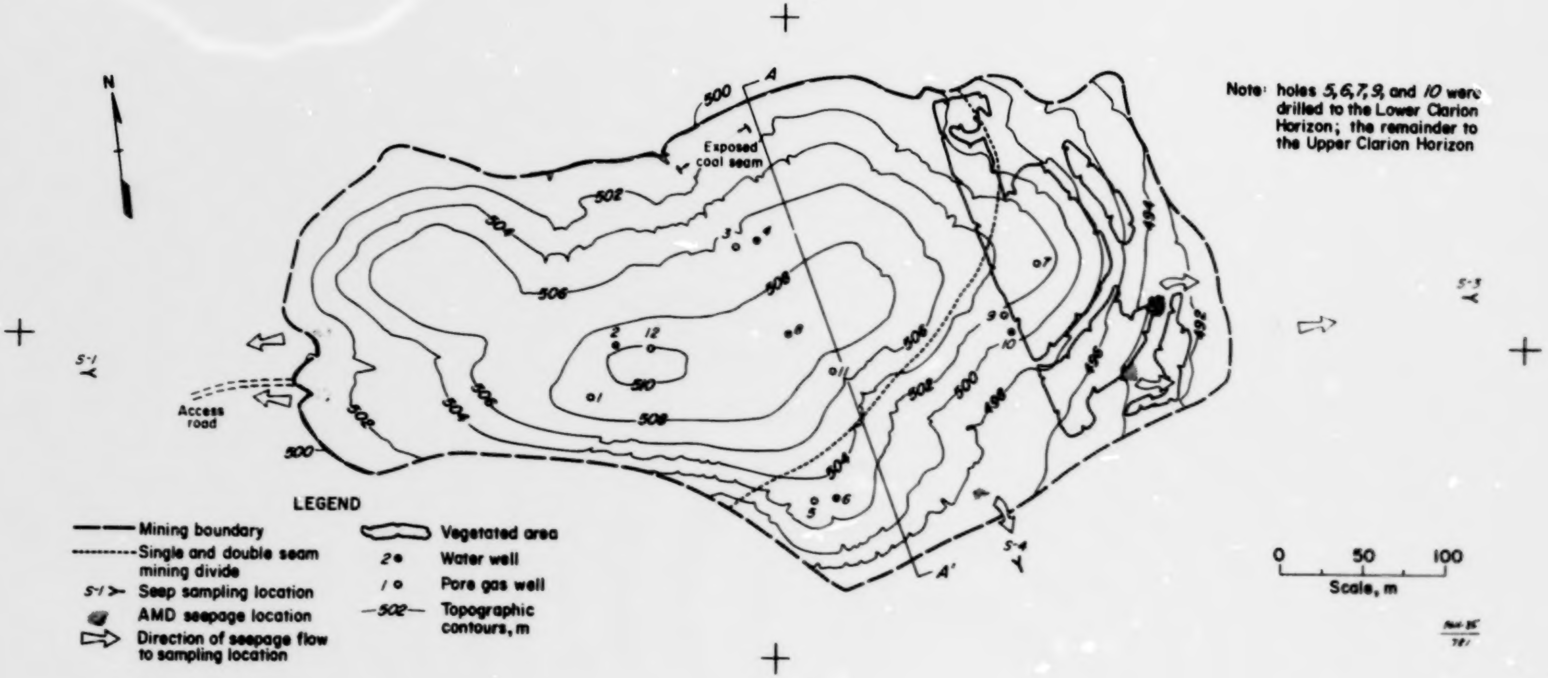
N
Note: holes 5,6,7,9, and 10 were drilled to the Lower Clarion Horizon; the remainder to the Upper Clarion Horizon
A
A'
500
Exposed coal seam
502
504
506
508
510
498
496
494
492
1
2
3
5
6
7
9
10
11
12
S-1
S-3
S-4
Access road
LEGEND
Mining boundary
Single and double seam mining divide
S-1 Seep sampling location
AMD seepage location
Direction of seepage flow to sampling location
Vegetated area
2 Water well
1 Pore gas well
502 Topographic contours, m
0
50
100
Scale, m

Figure 2. Map of the Clarion County site.

Table 2. Mean water quality for sampling stations at the Clarion County site, 1982-84 prior to revegetation.*

Sampling station	Acidity	Iron, total	Sulfate
Seeps			
S1	280.	9.0	750.
S3	710.	100.	1600.
S4	1400.	250.	2200.
Wells			
2	990.	160.	2700.
4	1500.	640.	2800.
6	390.	190.	940.
8	690.	210.	3900.
10	1200.	540.	2500.

*All concentrations in mg/L rounded to 2 significant figures; acidity reported as calcium carbonate equivalent.

Spoil analysis by the acid-base accounting method showed a generally moderate but variable acid potential, with more than half the samples showing less than 0.3 pct pyritic sulfur by weight. A number of samples, taken from various depths and boreholes, showed greater acid potential evidenced by pyritic sulfur contents in the range of 0.5 to 1.7 pct. Neutralization potentials were generally insignificant, but a few local values as high as 0.31 pct as calcium carbonate equivalent were found.

Oxygen profiles generally showed a consistent decrease in oxygen with depth. The steepest oxygen gradients were observed generally in the first sampling interval below surface (0 to 1.5 m), although steep gradients were also found at greater depth in several locations. Large decreases in oxygen content over a small depth interval are a clue that acid production may be occurring in the zone; however, because spoil gas diffusivity may vary with depth, other analyses must be used to confirm the acid source.

Data from water, gas, and spoil analyses were combined to develop an assessment of acid production for the Clarion County site. Pyrite oxidation appeared to occur for the most part within the uppermost 1.5 m of spoil. From oxygen profile measurements, the extent of oxidation within this zone appeared to be dependent on the pyrite content or the net neutralization potential of the spoil; however, we had insufficient data to test this hypothesis statistically. Oxygen content data also suggested the likelihood of acid production at greater depths when the near-surface spoil was depleted in pyrite, thus allowing the oxygen to migrate to an underlying pyritic zone.

The State attempted to abate acid production in conjunction with scheduled revegetation of this abandoned mine. Limestone was disked into the top 0.3 m of spoil at a rate of 327 mt/ha, followed by application of mulch and an acid-tolerant seed mix. The primary function of the alkaline addition was successful: the site has supported lush growth without soil addition since reclamation in 1984. However, only minor improvements have been observed in drainage quality to date.

Summary

These case studies illustrate common features of acid mine drainage from surface mines. Although the mines were operated approximately 10 and 20 years ago, contaminants are still being produced and discharged at high rates. Acid mine drainage is known to persist at many sites, including underground mines and coal refuse banks, for more than 50 years following closure.

In other respects, the mines represent very different characteristics. Drainage quality at the Upshur County site was dominated by a small zone of highly pyritic material. In contrast, acid production was disseminated across the Clarion County site. Despite a higher peak sulfur value at the younger Upshur County site, seepage from the older Clarion County site was much more heavily contaminated.

Combining data from the various monitoring methods allowed us to understand acid mine drainage at these sites. Water quality monitoring, of course, showed the lateral and temporal changes in contaminant formation and migration. Spoil analyses identified the potentially acid-producing material and pore gas analyses tentatively identified the vertical distribution of acid production. While initial acid abatement attempts were unsuccessful, improved understanding of such systems will allow the reclamation community to propose new means of controlling acid mine drainage.

Acknowledgments

Kenneth J. Ladwig and Peter J. Lusardi conducted much of the research described and authored the primary reports cited herein. Funding for the studies was provided by the Pennsylvania Department of Environmental Resources, the U.S. Office of Surface Mining through a grant to the West Virginia Department of Natural Resources and Acid Mine Drainage Technical Advisory Committee, and the U.S. Bureau of Mines.

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THE CAPE BRETON DEVELOPMENT CORPORATION'S VICTORIA JUNCTION COAL PREPARATION FACILITY A CASE HISTORY

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ABSTRACT

Acid Mine Drainage (AMD) is a problem confronting the coal industry in Nova Scotia. AMD has been with us since the first extraction of the mineral in the 1800's, but has only been appreciated for its impact on surface and ground water in the last 10 years or so.

This paper discusses a case history of acid mine drainage and coal mining in the Sydney Coal Field, specifically the holdings of the Cape Breton Development Corporation. The inherent problems associated with the generation of acid mine drainage from stockpiling coal and the wastes produced from coal beneficiation are discussed. Emphasis is placed on the impacts to ground and surface water environments adjacent to the Corporation's Victoria Junction Coal Preparation Facility.

INTRODUCTION

The Cape Breton Development Corporation was established by an Act of Parliament in 1967. The Coal Division of the Corporation acquired the operations of small companies operating in the Sydney Coal Field. The primary mandate of the Corporation at the time was to phase out the coal industry and establish new industry to sustain the economy of the Island. This direction changed with the energy crisis of the early 1970's. The Cape Breton Development Corporation, as we know it today, has rebounded with an ambitious investment of capital funds to build new facilities and maintain the older ones.

The Coal Division of the Cape Breton Development Corporation presently operates two submarine collieries on the northeastern shore of Cape Breton Island near Sydney, Nova Scotia (See Fig. 1).

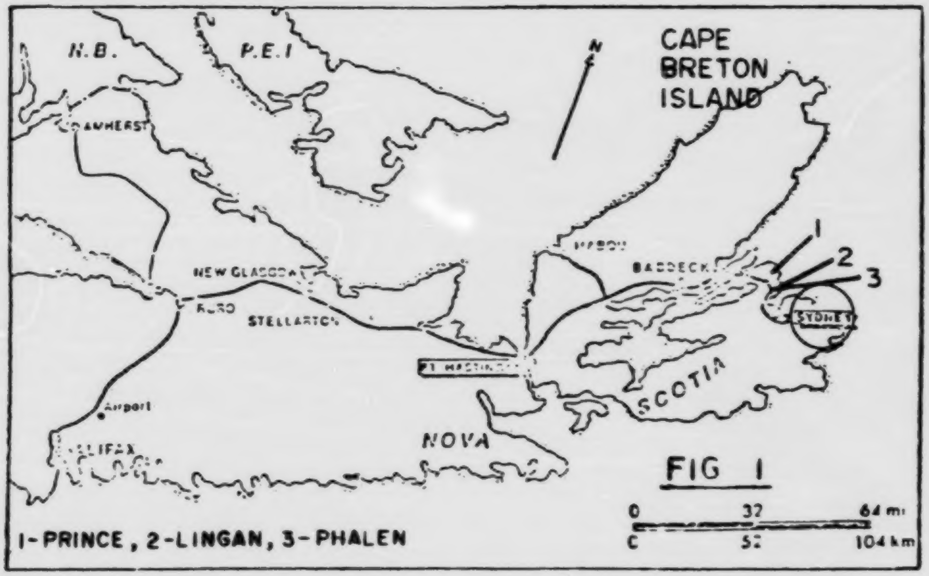
N.B.
P.E.I
N
CAPE BRETON ISLAND
AMHERST
NEW GLASGOW
STELLARTON
MABOU
BADDECK
1
2
3
SYDNEY
SCOTIA
NOVA
Airport
HALIFAX
0
32
64 mi
0
52
104 km

FIG 1

1-PRINCE, 2-LINGAN, 3-PHALEN

Lingan Colliery near the Town of New Waterford and Prince Colliery at Point Aconi, employ approximately 1900 men and produce between 2.3 and 2.5 million tonnes of saleable coal per year. Production is scheduled to commence at a third colliery, Phalen Colliery, in the spring of 1987. Phalen is situated adjacent to Lingan Colliery and will operate at a design capacity of 1.5 million tonnes of saleable coal annually.

Most of the Corporation's output from Lingan is beneficiated at their Victoria Junction Coal Preparation Plant, where two main product streams emerge. Thermal grade coal with an average sulphur concentration of 2.5% and ash at 12% is sold mainly for steam generation. The higher grade coal is sold for metallurgical processes at sulphur levels between 1.0 and 1.3% and ash at 2.5-3.5%. Coal in the Sydney Coal Field can be generally characterized as a high volatile A bituminous coal ranging in sulphur concentration from 0.7-6.0% and ash between 2-20% (DEVCO).

The Cape Breton Development Corporation's coal operation is supported by a corporate owned railway transportation system, fleet of tractor-trailer trucks, railway maintenance centre, preparation plant, shops and warehousing, as well as lifting and banking facilities at

Victoria Junction and a shipping terminal at Sydney Harbour. Over 3,500 employees are involved in the day-to-day operation of the Coal Division.

Once coal leaves the actual colliery site, it reports to the Victoria Junction Coal Preparation Plant and adjacent support facilities for beneficiation. There, coal is stockpiled, waste products are produced, saleable coal is banked and blended, and the largest potential in the Corporation for the generation of acid mine drainage exists.

The potential for AMD and its interaction with the sensitive environment adjacent to the Victoria Junction Facility is the subject of this paper.

THE VICTORIA JUNCTION SETTING

The Victoria Junction Facilities are located just north of Route 4 about 2 km east of the City of Sydney (Fig. 2). The site is bordered to the west by Grand Lake, a water supply for the Sydney Steel Corporation. Northwest Brook flows by the site to the south and east and is considered a valuable salmonid route. To the north the fine coal tailings basin, which receives fines from the preparation plant, lies adjacent to Kilkenny Lake, a potable water

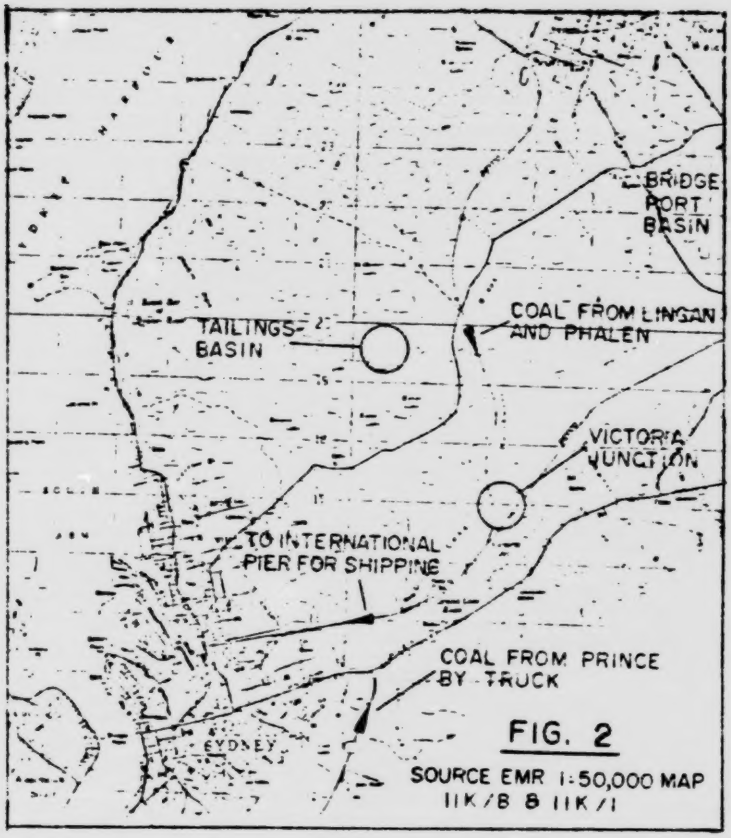
BRIDGEPORT BASIN
COAL FROM LINGAN AND PHALEN
TAILINGS BASIN
VICTORIA JUNCTION
TO INTERNATIONAL PIER FOR SHIPPING
COAL FROM PRINCE BY TRUCK
SYDNEY

FIG. 2

SOURCE EMR 1:50,000 MAP 11K/8 & 11K/1

supply for the Town of New Waterford. The entire site is underlain by a valuable Class I Bedrock aquifer (Nolan, Davis & Associates Ltd., 1983). All treated and untreated site discharges eventually make their way to Bridgeport Basin, considered a productive tidal inlet as well as a valuable resource for sport and recreation.

Operation of such a facility in this sensitive setting is not without its difficulties.

THE PROBLEM

The Cape Breton Development Corporation operates under the guidance of Provincial and Federal Environmental Agencies. As mentioned earlier, the energy crisis of the early 1970's triggered a new direction for the coal industry in Cape Breton. As a result, new facilties, including the Victoria Junction Coal Preparation Plant, materialized very quickly, perhaps much quicker than the technology necessary to safeguard against the inevitable impacts of acid mine drainage. Coincident with the increased activity in the Sydney Coal Field was the birth and development of the Nova Scotia Department of the Environment itself. It followed that the relationship between this new generation of coal mining and new environmental body was one of growing and learning together. When dealing with the inherent impacts of acid mine drainage, this association is still on the learning curve.

Through the years of investigations and environmental impact assessments at and around Victoria Junction, it has been determined that the most significant potential impact from the coal preparation cycle is the acidification of surface and groundwater flow systems.

Acid Mine Drainage problems are largely a result of ongrade storage of raw coal or washed product, but mainly storage of the fine and coarse waste products (E. D'appolonia Consulting Engineers Inc.). Seventy five percent of the course discard at the Victoria Junction Coal Preparation Plant is of the argillaceous type, or mudstone, siltstone and shale. This type of rock weathers rapidly exposing a large surface area to the elements. The other 25% is comprised of sandstone.

Climatically the Sydney coal Field falls in an environment that is characterized by average annual temperatures of 6°C, average annual wind speeds of 22.5 KPH, precipitation recorded with pH values as low as 4.6 and an annual water surplus of 293.9 mm (MacLeod, 1987). These conditions, coupled with the nature of the waste rock, appear to be conducive to oxidation of the pyrite, producing compounds that are both acidic and soluable in water. Percolation of this quality of water through the waste rock piles produces a leachate, or acid mine drainage, high in sulphuric acid and metal salts (E. D'appolonia Consulting Engineers Inc.)

The Victoria Junction Coal Preparation Plant has four locations which are particularly susceptible to acid mine drainage. They are designated in Fig. 3 as:

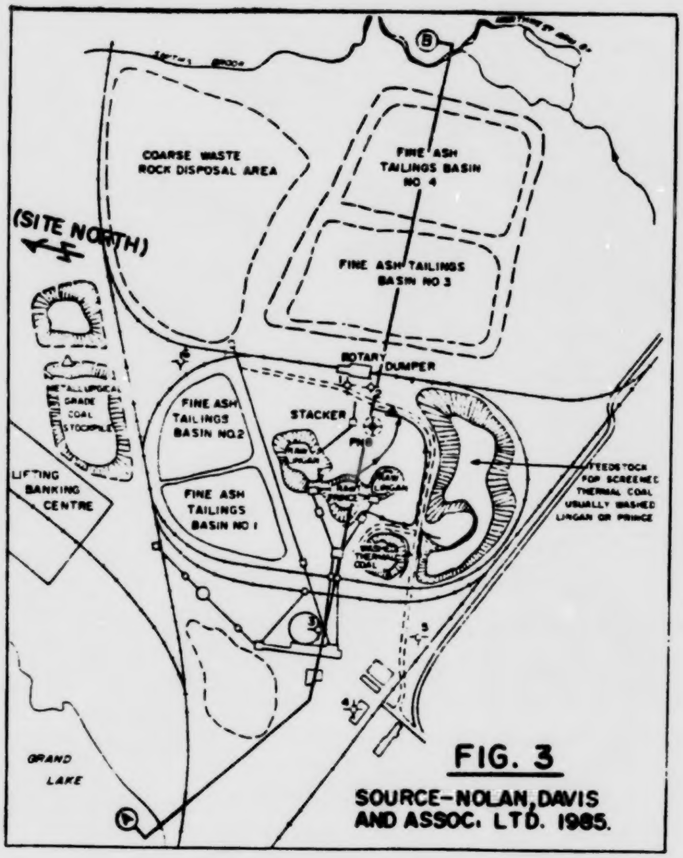
COARSE WASTE ROCK DISPOSAL AREA
FINE ASH TAILINGS BASIN NO 4
FINE ASH TAILINGS BASIN NO 3
(SITE NORTH)
ROTARY DUMPER
STACKER
METALLURGICAL GRADE COAL STOCKPILE
FINE ASH TAILINGS BASIN NO.2
FINE ASH TAILINGS BASIN NO 1
LIFTING BANKING CENTRE
PHB
RAW PRINCE
RAW LINGAN
WASHED THERMAL COAL
FEEDSTOCK FOR SCREENED THERMAL COAL USUALLY WASHED LINGAN OR PRINCE
GRAND LAKE
B
1
2
3
4
5

FIG. 3

SOURCE—NOLAN, DAVIS AND ASSOC. LTD. 1985.

.1 Raw Lingan, Raw Prince (Coal Feed to Plant)

.2 Washed Thermal, Metallurgical Grade Coal Stockpiles.

.3 Coarse Waste Disposal Area

.4 Fine Ash Tailings Basins (No. 1-4)

These areas are discussed below with their associated problems and how they are being handled.

THE EARLIER YEARS

When the preparation plant was commissioned in 1976, raw coal coming in from the collieries was stockpiled on grade ready to enter the preparation process. Coarse waste by-products were also transferred to an ongrade disposal area. Fine ash tailings were piped to holding basins for solids settlement and surface water decant. The saleable product was stockpiled on grade in two locations labelled thermal and metallurgical coal stockpiles in Fig. 3. Only decant water from the original Tailings Basin (Nos. 1 and 2) reported to an on-site treatment plant where lime was added to adjust pH and settlement capacity was supplied for metals. Runoff from the coal piles and coarse waste disposal area was virtually uncontrolled draining to the Northwest Brook watershed and eventually to Bridgeport Basin.

In 1980 the Nova Scotia Department of the Environment documented a contaminant plume beneath the Victoria Junction site. Nolan, Davis & Associates Ltd. (1983) attributed its origin to the ongrade storage of the fine tailings in ponds 1 and 2, as well as the coal and coarse waste storage piles. Noticeable signs of impact to Northwest Brook from surface water runoff was also evident. It was about this time that the Corporation began conducting intensive environmental impact assessments for new facilties and the engineering of mitigation measures for new designs to address the potential of acid mine drainage.

PROGRESS WITH AMD CONTROL TO DATE

In 1979, the Corporation took the first step towards properly managing its fine ash tailings circuit by retaining Geocon Inc. to investigate the feasibility of a large holding basin. The investigation resulted in the construction of a holding facility in a natural depression some 2.5 km north of the Victoria Junction site. Geotechnical and hydrogeological investigations determined that the facility would not impact on the water quality of the adjacent environments, including nearby Kilkenny Lake. The holding basin was developed by construction of an earthen dam on the downstream end of the depression basin.

Some of the mitigation measures incorporated include a sophisticated treatment and pH communication system where pH is read at three locations in the treatment system and telecommunicated back to the Victoria Junction control tower. A groundwater collection system was installed beneath the dam to pick up any contaminated groundwater, also reporting to the treatment plant. Following treatment, retention capacity is supplied for settlement of heavy metals.

Numerous groundwater monitoring wells were put in place around the basin to monitor the groundwater flow system for change. Groundwater movement in this area is such that detection of change in the monitoring wells will allow ample time for implementation of remediation measures.

Following two years of deposition, no change in groundwater chemistry has been detected. By keeping the majority of tails submerged in a reducing environment, treatment of the decant water has not been necessary to date.

For environmental and economic reasons the Corporation made a decision to construct a lifting and banking area (LBC) to handle the problems associated with ongrade storage of the saleable product. In early 1983 an environmental impact assessment was conducted on a plot of land adjacent and to the north of the Victoria Junction Site (See Fig. 4). Development of the LBC started late in the same year.

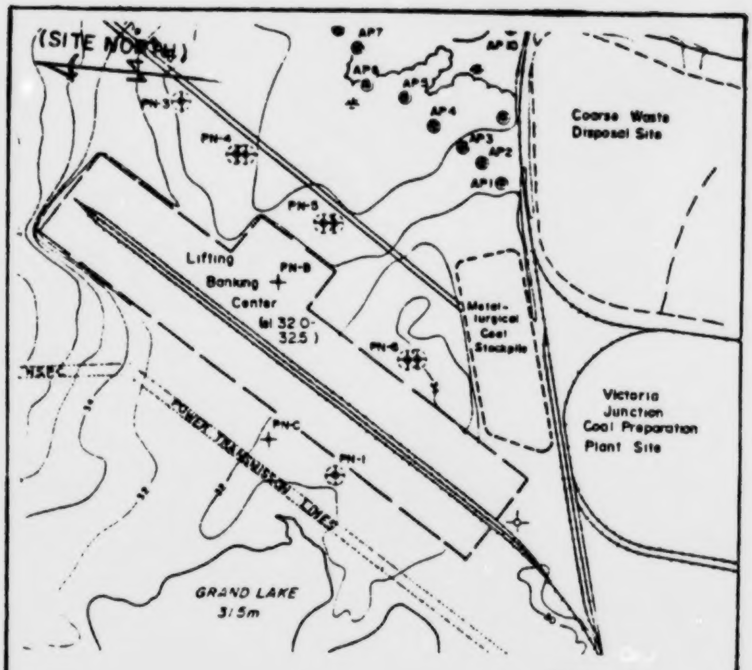
(SITE NORTH)
AP7
AP10
AP6
AP5
AP4
AP3
AP2
AP1
Coarse Waste Disposal Site
PN-3
PN-4
PN-5
PN-B
Lifting Banking Center
(el 32.0-32.5)
PN-6
Metallurgical Coal Stockpile
Victoria Junction Coal Preparation Plant Site
PN-C
PN-1
POWER TRANSMISSION LINES
GRAND LAKE
315m

FIG. 4

SOURCE - NOLAN, DAVIS & ASSOC. 1985

The findings of the impact asssessment were closely addressed during the construction of the facility. By using the Calcite Saturation Index for groundwater, Nolan, Davis & Associates Ltd. (1983) showed that the groundwater flow system in the area of the proposed LBC has a low sensitivity to acidification. It followed that a "delay and decay" management philosophy (i.e., allow the groundwater flow system to attenuate any contaminant plume before exiting a predetermined buffer zone) for any contaminant plume that may develop beneath the site was acceptable to the regulatory agencies.

The facility has a 12.5 cm paved subfloor and ditch system to minimize percolation of low pH pile drainage to the surface and groundwater flow systems. The drainage reports to an on site settling pond for removal of solids and the decant is piped off site for eventual treatment at the Tailings Basin.

Based on this management philosophy, the groundwater flow system was studied in greater detail and a monitoring program designed and installed to test it (Nolan, Davis & Associates Ltd., 1985).

The Tailings Basin and Lifting and Banking Centre represent a Corporate effort to safeguard and implement mitigative measures necessary to avoid future impact from acid drainage. The Corporation is also involved with remediation measures to correct control problems resulting from the earlier days when few controls were installed.

A bentonite/clay slurry cutoff wall is being installed on the east and south sides of the Victoria Junction coarse waste disposal area and tailings basins no. 3 and 4 (See Fig. 5). The cutoff is intended to influence surface drainage from the area, as well as contaminated groundwater flowing in the basal till, to drain to a collection facility. Construction of the cutoff was started in

November, 1986 but was suspended due to weather constraints. The project will commence in the spring of 1987.

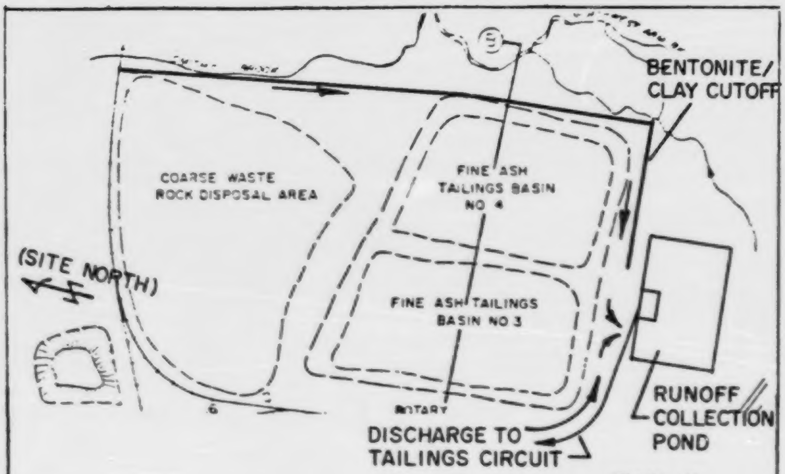
BENTONITE/
CLAY CUTOFF
COARSE WASTE
ROCK DISPOSAL AREA
FINE ASH
TAILINGS BASIN
NO 4
(SITE NORTH)
FINE ASH TAILINGS
BASIN NO 3
ROTARY
6
RUNOFF
COLLECTION
POND
DISCHARGE TO
TAILINGS CIRCUIT

FIG. 5

SOURCE-NOLAN, DAVIS & ASSOC. 1985

The small onsite treatment facility was recently converted to a pumping station with a pond holding capacity of 16,000 m^3. All site runoff, including that from the coarse bank area, reports to this station. Two pumps capable of 130 LPS transport low pH runoff to the fine tailings circuit where it is used as make up water for the transmission of the tailings to the basin. Final discharge following treatment at the basin is to the Northwest Brook trunk via a tributary.

A very extensive groundwater monitoring system has been installed to delineate the contaminant plume from the waste pile and tailings ponds. Following completion of the cutoff wall this system will serve to test the effectiveness of the remediation measures.

Looking to the future, a long range management strategy aimed at controlling acid mine drainage from the coarse refuse area was put together for the Corporation by Monenco Inc. (1985). The reclamation strategy is already underway utilizing a phased approach whereby the grade of the bank slopes have been reduced to 3 to 1 from an angle of repose of approximately 1.5 to 1. Deposition will continue for the remaining life of the pile along these grades. A test plot experiment will commence in the spring of 1987 to determine the optimum ammendments and vegetation to seal the pile from the elements most responsible for acid water generation, water and air.

THE FUTURE

The Victoria Junction Coal Preparation Plant is currently undergoing an expansion which will essentially double its output. Additional fine coal recovery circuits are being installed to improve the efficiency of the plant by reducing the amount of fine coal lost to the fines waste stream. This should, in effect, reduce the potential for AMD at the Tailings Basin.

Corporate efforts in the field of coal-water fuel production technology is gaining momentum. This technology will allow the utilization of poorer quality coals resulting in a net reduction of the potential for AMD from the present method of coal handling.

Remedial measures now underway to control AMD from the coarse waste disposal area and to handle site runoff will be completed in the spring. The success with mitigation measures taken during construction of new facilties, must be borne out through continued monitoring of the flow systems.

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ACID MINE DRAINAGE IN REACTIVE SLATES

"THE HALIFAX AIRPORT CASE"

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ABSTRACT

Since 1960, there have been several fish kills and other documented environmental impacts related to acid drainage from the Halifax International Airport and environs. Environment Canada has actively worked with all parties, notably Transport Canada to resolve this environmental problem.

We are encouraged by recent events but our experience suggests that constant vigilance and further efforts are necessary to ensure that additional degradation does not occur.

The fish kills in the Shubenacadie River in the fall of 1960 were the first highly visible sign that some human activity in this area was impacting on this environment. Several investigators over the ensuing years determined that the construction of an airport and highway, through a highly fractured fissile band of mineralized slate, had indeed resulted in acid drainage which was believed to have caused the fish kills. This "acid", typically released during heavy rainfall, usually following a prolonged dry spell, depressed the pH and increased levels of heavy metals in the watersheds surrounding the airport, particularly Bennery Brook.

The airport is situated on a drainage divide between the Bay of Fundy and the Atlantic Ocean. About 64% and 20% of the Airport property drains to Bennery Brook and the Johnson River respectively. Both discharge to the Shubenacadie River which then empties into the Bay of Fundy. The remainder (16%) of the Airport drains to the Fish Hawk Nest River and then to the Salmon River and the Atlantic Ocean via several headwater streams. Because of the widespread disruption (and uncontrolled use) of slates on the airport property and highway construction, the acid generation was not restricted to a few well defined sites. As a consequence, all receiving watersheds were and continue to be affected.

The series of other fish kills over the next 20 years or so demonstrated that over and above any "baseline" impact from initial development, there could be significant incremental impact from ongoing activities. This suggested that the environment, while considerably stressed already, is really dramatically affected by additional impact. This realization would, eventually, force us to reconsider our whole approach to the problem.

While it had been determined at a very early stage that disturbing and/or exposing this mineralized slate invariably promoted the generation of acid, this did not entirely eliminate (or even much slow down) activity in the slate over the next two decades. Expansion and growth of the airport brought continual pressure to develop. Unfortunately, the logical areas for this growth often coincided with the slate zones. Each of these extensions caused varying degrees of slate disruption and corresponding acid generation.

In retrospect, Environment Canada (or other environmental agencies) certainly didn't have its act together through this period, to the point where we much influenced the activity of Transport Canada, or others. In fact, the biggest concern through these early years seems to have been the rapid deterioration of iron pipes used for transporting sewage to a treatment facility and for deterioration of sub drainage systems. Other than knowing that there were the occasional fish kills, we did not have a good appreciation of the problem. In fact, we didn't really know, in a quantitative sense, what the scope and breadth of the problem really was. We all continued to invariably underestimate the degree of acid generation associated with normal construction activity and to overemphasize the anticipated return on minimal mitigative measures, such as a 15 cm layer of clay cover would provide.

It wasn't until the late 1970's that any serious attempt was made to integrate the investigations to the point where the larger picture started to emerge. The Environmental

Clean-Up Program Assessment Study by Canadian British Consultants Limited (CBCL) in 1978 identified the major contributors to the acid generation, from the airport property, and also recommended mitigation measures.

Shortly after this study was completed, Environment Canada carried out an internal evaluation of the control measures recommended by CBCL. The major finding of this study, was that the airport was not the major contributor to acid generation to Bennery Brook! Two external quarries, in fact, were found to be significant sources. This clearly told us that mitigative work at the airport alone would not solve the environmental problem. Much dialogue was then conducted amongst Environment Canada, the provincial departments of Transportation & Communications and Environment and of course, Transport Canada over the next several years. Still however, no real gains were made in dealing with the problem.

As John indicated, the one event that really was to bring the whole problem very sharply in focus was the new taxiway constructed in 1982. The created waste rock pile, along with the taxiway itself, soon caused quite remarkable acid generation and severely impacted the Fish Hawk Nest River, Salmon River - Lake Echo system.

The significance of this project was manyfold. It was certainly the first time that Transport Canada was required to treat the acidic waters, in a physical sense. Despite attempts to cover the exposed slate and intercept surface waters, extreme acid generation continues to this day. This, in turn gave Environment Canada grave concern for the ability to control the generation of acid by conventional cover/bury methods, unless the maximum effort is expended.

At about the same time, the County of Halifax announced that they planned to develop an industrial park in an area immediately adjacent to the airport, draining towards Miller & Soldiers Lake.

At this point then the environmental agencies found themselves at a very crucial juncture. We really believed the battle would be lost if we continued on a "catch if catch can" approach-eventually the environment would succumb to the cumulative impacts. It was apparent that if left unchecked, the airport and environs was definitely going to experience rapid growth, some of which would occur in the slate. We obviously either had to establish a moratorium on all future development in slate or define groundrules for controlled development (and at the same time work on mitigative measures for past activities). Realistically, we chose the latter but through administrative measures also essentially effected a short-term moratorium until rules could be established to allow for controlled development.

In order to achieve this, we established a committee, bringing together for the first time, in a structured way, participants from the Nova Scotia Department of Environment, and Environment Canada who could contribute to the development of a policy. I am pleased to say this committee has been successful in developing a draft policy which has already been extremely useful in removing most of the uncertainty for development. I have brought along copies of this draft policy and can provide additional ones if anyone is interested. I feel compelled to point out that the overriding theme of this policy remains to minimize disturbing the slate at all costs despite the establishment of criteria when slate is disturbed.

As an adjunct to this, we knew that we would eventually have to define the environmental boundaries - acceptable environmental quality and associated with this acceptable effluent quality standards. This obligated us to get a much better data base from which decisions could be made. In other words get out into the field and validate some of the postulations and theories. This we did and continue to do. It has proven very beneficial. We have just, as a consequence, completed development of these discharge criteria and I have also brought copies for distribution.

As well, some research has been supported into the prediction of acid generation and the evaluation (assessment) of slate related problems in other areas of the province. This we believe will eventually allow us to prioritize slate areas either for development, (or exclusion) or use as a quarry, and will add greatly to our confidence in allowing or denying this activity.

Lastly, we quickly saw the truth in the old axiom "The more you know, the more you realize how little you know!. We believed that others would also be interested in sharing their experience and knowledge with us. We therefore embarked on having this workshop and were quite overwhelmed and pleased about the response. I can assure you that what we learn here will unquestionably influence and modify our future strategies.

I know that you will hear much more about this particular problem in later presentations, so I will not deal with it any more specificially here. I would in closing however like to say a few words about our "experience" in dealing with this problem.

From our perspective it has often been very frustrating. Gains where we have realized them, have been painful to exact and often have seemed minor alongside the losses. We have often felt like we were going one step forward and two steps back. We have concluded that constant attention and diligence must be exacted from all parties, including ourselves. For example, random audits and checks have not been found to be adequate. As well, for the control strategy to work, it must be applied fairly and universally.

We have convinced ourselves that all activity should start and end by minimizing disruption of the slate. This point cannot be overemphasized. Should it be necessary to disrupt the slate, immediate and complete mitigation measures must be taken. Half measures are considered totally ineffective.

We also realized that we must provide everybody with clearly articulated environmental guidelines. We believe we have started this process but recognize that improvements can be made.

Lastly, we have learned that the environment is remarkably resilient despite our best efforts to destroy it. That fish kills occurred over a long period of time is indicative, among other things, that given half a chance, the streams can become repopulated. Obviously, it is up to us all to do our part to see the environment gets this opportunity.

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Acid Mine Drainage in Reactive Slates "The Halifax International Airport Case" - The Transport Canada Perspective

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Abstract: This paper outlines Transport Canada's past experience with acid mine drainage at Halifax International Airport and what is now being done to control this problem. A brief summary of the lessons learned by Transport Canada is given.

The Halifax International Airport is located on 972 hectares of land, in the County of Halifax, thirty-seven kilometres north of the provincial capital, Halifax and the City of Dartmouth. The land adjacent to the airport is sparsely settled, the closest communities being Enfield, 8 kilometres north, and Waverley 11 kilometres to the southwest.

Halifax International Airport is built on a 2½ km wide band of slate which is highly fractured and contains pyrite and other minerals including Al, Fe, Mn, Co, Ni, Pb and As in varying concentrations as shown in Table 1.

Table 1

Element	Percentage	ppm
Aluminum	9-13	
Iron	3-6	
Manganese	.01-.04	
Cobalt		10-80
Nickel		40-270
Lead		20-48
Arsenic		9-11

The bedrock when exposed either naturally or as a result of blasting or excavation generates acid mine drainage.

In 1955, the City of Halifax expropriated the site for the present Halifax International Airport and the Department of Transport commenced construction of the airport in November of that year. The airport was opened in July 1960 and is owned and operated by Transport Canada.

The construction of a new Provincial highway passing through the Airport property was carried out concurrently with the construction of the Airport. This road has since been upgraded to a 4 lane divided highway and an interchange was added to service traffic to and from the Airport.

The reference elevation of the airport is 145 metres above sea level. With the exception of a few small projections, the airport is the highest point in the surrounding area.

The terrain generally slopes away from the site and consists of numerous small lakes and rolling hills. Drainage thus originates on Airport property as several headwater streams.

The Airport is situated on the drainage divide between the Bay of Fundy and the Atlantic Ocean.

During the late summer of 1960 the first of several major fish kills occurred in the Shubenacadie River. These fish kills were attributed to the construction of the Airport and Bicentennial highway by the Province which produced acid mine drainage.

Since 1960 several major fish kills have been recorded. These kills and degradation in water quality are well correlated with major construction projects in the area (See Table 2).

Table 2

FISH KILL HISTORY IN THE SHUBENACADIE RIVER

(From EPS, 1976)

MAJOR CONSTRUCTION		MAJOR FISH KILLS
Terminal and Runways	1957-1960	September, 1960
Imperial Oil	1959-1960	
IMP Hangar (large)	1959	
IMP Hangar (small)	1961	
Air Canada Hangar	1961	
Air Halifax Hangar	1961	October 1961
Highways slate needs	1965	October, 1965
Avis Service Station	1966	September, 1966
Highways slate needs	1968	November 1968
Halifax Flying Club	1970	
Mobil Oil Hangar	1972	
Highway Overpass	1974	August 1974
Aircon Tank Farm	1975	October 1975
EPA Hangar	1976	September 1976

The Shubenacadie River is also used as a potable water source for a number of small communities downstream of the Airport.

During the planning, design and construction of the Airport, environmental concerns such as acid mine drainage were not given much weight. Had acid mine drainage been considered during the original siting of the airport, the location chosen would likely have been different. During initial construction, pyritic slate was used under the aprons and runways as a base material. It was also used as a base for the Bicentennial highway constructed by the Province.

Even as late as 1974 pyritic slate was not a concern as evidenced by the lack of mention of pyritic slate in the Airport Master Plan which covered the 20 year planning period 1973-93.

An environmental study carried out in 1976 documented the environmental impacts from acid mine drainage and recommended that all exposed slate be covered with about 3 feet of compacted clay covered with soil and either seeded or sodded. A lime treatment facility for some of the airport waters was also recommended. These recommendations were not implemented.

In 1982, the construction of a parallel taxiway which required the excavation of 225,000 cubic meters of pyritic slate bedrock brought the problem of acid drainage to a head. Some of this bedrock was used for taxiway fill and some was placed in a 7 ha disposal site adjacent to the taxiway which is referred to as the waste pile. Drainage from the taxiway and waste rock pile discharges via headwater streams to the Salmon River watershed. After taxiway construction, the taxiway underdrains and waste rock pile began to discharge large quantities of low pH and heavy metal laden water.

In response, Transport Canada constructed a temporary lime treatment facility at a cost of $500K designed to neutralize the acid drainage and to prevent environmental damage to the receiving Salmon River watershed. Although the treatment system generally achieves an effluent quality sufficient for surface stream discharge, it experiences reduced treatment efficiency under winter conditions and handles only approximately one-half of the total acid drainage discharged to the Salmon River due to underdrain and waste rock pile bypass. Furthermore, a significant disposal problem has developed as a result of the massive accumulation (26,000 cubic metres) of lime sludge.

Capping the waste rock pile with 750 mm of clay and 150 mm of topsoil at a cost of $800K is nearly completed. Seeding shall be done this spring. This clay cap will significantly reduce the amount of acid mine drainage that requires treatment.

Other costs associated with acid drainage at this site include $240K annually for the operation and maintenance of the treatment plant. This temporary plant is reaching the end of its useful life and will have to be replaced or upgraded unless significant improvements are noted in water quality following the capping of the pile.

A stormwater management plan is underway to identify possible permanent solutions to acid drainage.

An earthworks logistics plan is also underway to identify major sources of acid drainage at the site and recommend corrective action on a priorized basis. These two studies will cost about $200K.

Dewatering and solidification of the accumulated sludge is expected to begin this year using conventional mechanical technologies.

Future

All future development at Halifax International Airport will have to incorporate measures such as above ground construction, covering exposed slate within a prescribed time etc. These constraints will significantly increase the cost of developing land at the Airport for tenants and Transport Canada.

Environmental considerations must be included in the early planning stages of all projects involving the disturbance of slate to ensure that funds for environmental protection are included in the total project cost.

All construction projects at the Airport now require an Initial Environmental Evaluation (IEE) which is reviewed by Environment Canada to ensure environmental protection in the early stages of the planning cycle.

Lessons Learned

If we were to plan Halifax International or any other Airport today there would be a requirement to evaluate, as part of our site selection process and as a result of the Federal Environmental Assessment and Review Process, the choice of alternative sites from an environmental point of view. This evaluation would identify potential problems from acid mine drainage at the present site. The costs and restrictions associated with pollution contol at this site would be considered in the planning process. A different site would likely have been chosen.

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ACID DRAINAGE FROM MINERALIZED SLATE AT HALIFAX AIRPORT

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ABSTRACT

In January 1982, approximately 300 000 $yd.^3$ of excavated slate at the Halifax International Airport was placed at a disposal site (waste rock pile) situated in the watershed of the Salmon River. When acid drainage from the site was identified in June 1982, lime addition and settling (ponds) were initiated as temporary treatment. Contrary to earlier thought, acid drainage began increasing in strength, overloading the treatment facility.

Unpredictably large volumes of acid drainage were generated by chemical oxidation in the anaerobic saturated lower layers of the waste pile and fractured bedrock in the excavated area of the new taxiway. Thiobacillus ferroxidans acted as the catalyst to initiate and sustain the chemical reactions.

Four years of water quality record has revealed that acid drainage production in the waste rock pile and fractured bedrock has reached an apparent peak in late 1984-1985, with lower dissolved metals and sulphate reported in 1986. The acid drainage discharged from the fractured bedrock in the underdrains is highly variable in strength related to seasonal climatic variances and precipitation events.

There is vertical movement of acid drainage from the waste rock to the underlying bedrock and possible subsequent discharge to the swamp northwest of the settling ponds. Accompanying this movement is dissolution of the glacial till and additional release of metals.

The inherent structural and mineralogical characteristics of slate at the Airport demonstrates the susceptibility of the rock to strong acid production. The low grade of metamorphism results in a high degree of cleavage and high mobility of rock components, particularly leachability of metals. This along with a high degree of pyrite crystal surface area, makes the Halifax Airport situation a unique one from a global perspective, matching acid drainage strengths published in the literature.

INTRODUCTION

Acid drainage has been a serious problem at mine tailing sites for many years. The low pH and high concentrations of metals in acid runoff have caused significant toxic effects on biota of receiving waters.

In Nova Scotia, acid drainage is generated at construction sites from exposed sulphide mineralized bedrock of the Halifax Formation slates. These slates stretch for a width of approximately 50 kilometres across the entire southern region of the Province. Many streams and lakes are presently highly stressed from acid drainage runoff. At the Halifax International Airport, several fish kills have been attributed to acid drainage runoff from construction activities between 1969 and 1978 (E.P.S. 1976, Thompson 1978).

In 1982, about 300 000 yd.^{3} of highly mineralized slate bedrock was blasted and removed to accommodate the installation of Taxiways B and D, commonly referred to as Delta (Figure 1). The bedrock was used as taxiway fill with the remainder placed at a 17 acre disposal area referred to as the waste rock pile and capped with 2 feet of clayey glacial till. The bedrock cut area was approximately 460 feet length, 100 feet in width and 15 feet in depth.

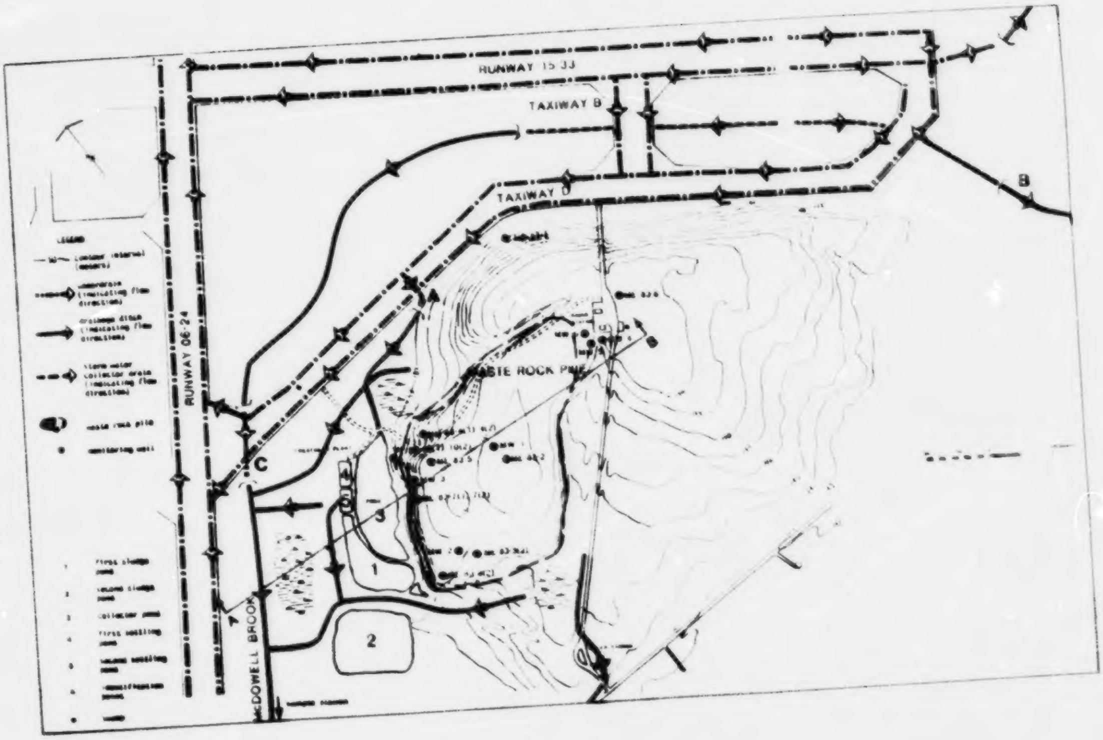
RUNWAY 15-33
TAXIWAY B
TAXIWAY D
RUNWAY 06-24
WASTE ROCK PILE
McDOWELL BROOK
A
B
C
1
2
3
waste rock pile
monitoring well

Figure 1

An underdrain system of conduits was placed beneath and alongside the taxiways to depress the water table in order to maintain the integrity of the paved taxiway surface. A storm water collector system was also constructed with piping placed below ground surface between Runway 15-33 and Taxiway D. The discharge from these two systems is located at Stations A, B and C (Figure 1).

Within two months following the completion of construction, acid drainage began discharging (at pH 3) from the underdrain/stormwater systems and base of the waste rock pile. A lime treatment facility was established to neutralize the runoff prior to discharge into McDowell Brook, which is a headwater stream of the Salmon River (Figure 1). The acid runoff from the waste rock pile and Station A was directed into a collector pond and then pumped into neutralization tanks where hydrated lime slurry was added. The neutralized slurry was then discharged to two settling ponds of 1,300 $yd.^3$ capacity each, for removal of the precipitated solids in the form of a gypsum (metal) hydroxide sludge. The supernatant at pH 9-11 was allowed to free discharge from the second settling pond to McDowell Brook. The storm water collector discharge at Stations B and C was allowed to discharge, untreated, to McDowell Brook and Leech Brook (another headwater stream of Salmon River).

The metal hydroxide sludge is pumped to storage in two sludge ponds of 13,000 $yd.^3$ and 22,000 $yd.^3$ capacity. An estimated 20,000 to 26,000 $yd.^3$ of sludge has accumulated as of early 1985 (Porter Dillon, 1985).

The quality of the acid runoff and water in McDowell Brook was continuously monitored to evaluate the treatment effectiveness. The general trend of thought was that acid drainage would cease within two years and the treatment plant would then be closed. After two years of treatment, however, the acid runoff increased in strength, with lower pH values and higher concentrations of dissolved metals reported (Porter Dillon, 1985).

This led to the initiation of this project in early 1984 in cooperation with the Centre for Water Resources Studies at the Technical University of Nova Scotia and Transport Canada. The purpose of this study was to evaluate the chemical pathways involved in acid drainage generation from the Halifax Slates, and determine the time frame involved for continued generation.

METHODS

Ground water monitors were established in the waste rock pile and around the new taxiway to enable ground water sampling and determination of ground water flow characteristics (Figure 1). Three 1.25 inch diameter bedrock cores were taken in 10 foot lengths along an exposed bedrock area south of the Radar Station (MW's 4, 5 and 6 Figure 1). Twelve (12) waste rock and six bedrock samples were analyzed for acid producing and consuming potentials, percent sulphur and metals (aluminum, iron, manganese, copper, lead, zinc, nickel, cobalt, cadmium and arsenic). The acid producing tests were conducted in accordance with standard leach tests established by GSC (Duncan, 1972). With regards to metals, the whole rock method was used to determine the total percent of the element in the sample and the acid leach method was used to determine the leachable portion of metals contained in the sample.

Five waste rock samples were submitted to the Pathological Institute for determination of the presence of the bacteria *Thiobaccilus ferroxidans*.

Water sampling from the piezometers began in April 1984 and is ongoing on a regular basis. Samples were collected with a portable PVC bailer following purging of three well volumes from the monitors. All samples were filtered immediately upon arrival at the laboratory and samples separated for metal scan analysis were acidified with dilute nitric acid within 12 hours of sampling.

RESULTS

Bedrock Geology

The Halifax Formation is comprised mainly of slate and meta-greywacke with minor quartzite. The rock is typically ferruginous (iron-bearing) and aluminous. The slates of the Halifax Formation are markedly enriched in sulphides, many times greater than quartzites and greywackes. Visible sulphide mineralization occurs as dessiminated crystals throughout the host rock and concentrated along cleavage planes and quartz veins (Keppie, 1983). Sulphide minerals of pyrite and arsenopyrite with minor amounts of pyrrhotite, sphalerite and chalcopyrite were observed in the core at the Airport. These minerals are the principal source of acid drainage at the Airport and contain metals of iron, arsenic, copper, zinc, nickel and aluminum. Laboratory analyses (whole rock method) indicate iron concentrations of 3-6%, aluminum 9-13% and arsenic 10 ppm (Table 1).

Over the past several million years, the exposed sulphide minerals have weathered to a stable oxidized form generally viewed as reddish brown iron oxide coating on these mineral grains or have leached from the rock as evidenced by a pitted bedrock surface common in the Airport area. It is the unexposed minerals disseminated in the host rock and concentrated along quartz veins and cleavage planes which form the source of acid drainage. Blasting causes the bedrock to

TABLE 1
GEOCHEMICAL WHOLE ROCK ANALYSIS

			ppm							%		
Rock Type	Sample	Depth (ft)	Cu	Pb	Zn	Ni	Co	Cd	As	Al	Fe	Mn
vein in quartzite	TH#4	2.0 - 2.2	33	38	65	40	16	<.5	9	12.57	3.5	.013
quartzite		4.0 - 4.2	40	20	210	68	24	<.5	9	9.93	5.45	.04
waste rock	TH#1	20	43	28	136	69	28	<.5	19	11.20	4.34	.03
slate	TH#5	2.0 - 2.2	27	40	90	44	10	<.5	10	12.35	3.05	.021
slate		3.0 - 3.2 1	46	48	133	270	80	3.0	11	11.23	6.10	.012
waste rock	TH#2	25 - 27	18	10	73	40	12	<.5	18	11.78	3.14	.024
waste rock	TH#3	10	32	18	107	58	14	<.5	4	10.64	3.0	.03
		20	26	22	130	40	13	<.5	16	11.66	3.05	.028
		25	30	16	104	44	14	<.5	7	10.92	3.14	.028

break along these cleavage planes, essentially shattering the rock, particularly slate. This greatly increases the surface area exposure of the rock and the associated sulphide minerals.

The ability for Halifax Slates to fracture is very well pronounced at the Halifax Airport where the cleavage is very well defined because of its low metamorphic grade (Graves, 1986, personal communication).

The metamorphic geology has a significant effect on the acid producing capacity of the bedrock. The regional metamorphic grade increases systematically from the Airport where the Meguma rocks are as low grade as any in Nova Scotia to both the eastern and western ends of Nova Scotia where the grade is considerably higher (Muecke and Keppie, 1979). The bulk effect of this gradient is to decrease the amount of CO_2 and increase the relative amount of SiO_2, thus changing carbonate minerals to silicate minerals. The rock becomes less porous, harder, and the cations more tightly bound into coarser-grained metamorphic minerals (Purves, 1984). We might expect, therefore, the mobility of rock components as well as cleavage) to be greatest in areas of lowest metamorphic grade, such as at the Airport (Graves, 1986, personal communication). The acid leach tests indicate that approximately 84-88 percent of the total available iron is potentially leachable from the rock (Table 2) and only 13-18 percent of the aluminum is leachable (the remaining aluminum tightly bonded as aluminum silicates).

TABLE 2
ACID LEACH ANALYSIS

			ppm							%		
Rock Type	Sample	Depth (Ft)	Cu	Pb	Zn	Ni	Co	Cd	As	Al	Fe	Mn
waste rock	TH#1	10	28	14	64	13	5	<.5	8.	1.26	2.97	.066
		20	46	28	116	51	26	<.5	2.6	1.51	3.73	.032
		25	41	18	86	35	22	<.5	4.8	1.33	3.32	.042
waste rock	TH#2	10	32	20	76	20	8	<.5	15.0	1.39	4.53	.065
		20 - 22	36	16	71	16	6	<.5	12.8	1.25	4.24	.022
		25 - 27	19	20	65	21	14	<.5	8.6	1.30	2.78	.024
		27 - 27.5	20	18	70	23	9	<.5	8.6	1.20	2.82	.023
quartzite	TH#4	4.0 - 4.2	47	24	196	59	25	<.5	3.0	1.79	4.81	.036
slate	TH#5	2.0 - 2.2	28	22	75	38	13	<.5	0.5	1.55	2.54	.024

Rock disaggregation is well evidenced by the material contained within the waste rock pile which is now, in places, a fine black powder with surface area exposure essentially approaching 100 percent (Lund, 1985).

The available surface area of the mineral is also important. Results of experiments conducted by a Ferguson (1985) indicate that there is a geometric relationship between acidic production with total exposed surface area. In general, acidic production was shown to increase dramatically at a mean particle size less than approximately 10 mm. This may explain why fresh bedrock collected at the Airport containing disseminated small particles had higher acid producing capabilities than pyrite concentrated as large crystals along quartz veins. Acid producing potentials are in the range of 13-157 kg/tonne of rock and acid consuming potentials in the range of 3.4-6.3 kg/tonne (Table 3).

TABLE 3

ACID PRODUCING POTENTIAL AND CONSUMING ABILITY

Rock Type	Sample	Depth (ft)	Percent Sulphur	Acid Producing Potential (kg/tonne)	Acid Consuming Potential (kg/tonne)
vein in quartzite	TH#4	2.0 - 2.2	1.32	40.7	4.9
slate	TH#5		5.10	157.1	3.4
waste rock	TH#3	10	0.65	20.0	3.5
		20	0.3	9.3	3.3
		25	0.296	9.1	3.0
waste rock	TH#1	10	0.22	32.8	4.3
		20	1.21	37.3	5.0
		25	0.88	27.1	5.3
waste rock	TH#2	10	0.27	8.3	2.5
		20 - 22	0.29	8.9	1.9
		25 - 27	0.72	22.2	3.1
		27 - 27.5	0.32	9.9	2.6
quartzite	TH#4	4.0 - 4.2	1.32	40.6	4.7
slate	TH#5	2.0 - 2.2	0.45	13.8	6.3

Results of the leaching tests suggest that 30 to 60 percent of the metals (with the exception of arsenic) have leached from the waste rock pile in two years. This observation is, however, highly subjective based on extreme variability of metal concentrations and the limited number of samples. The range of values within the waste rock pile, nonetheless, are distinctly lower than those observed in the rock samples (Table 2).

Water Quality

Water quality in undisturbed bedrock ground water (represented by data collected in 1983 in ME-83-6) reflects a calcium sulphate type water with basic pH, high alkalinity and low acidity (Table 4). Metal concentrations are below detectable limits, except iron and manganese.

TABLE 4
WATER QUALITY RESULTS
(all units reported in mg/L)

	Bedrock Undisturbed	Bedrock Disturbed	Taxiway Underdrain	Stormwater Interceptor	Waste Rock Pile	Waste Rock Seepage
Na	4 - 25	3.1 - 7.5	-	-	0.6 - 33	2.0 - 9.8
Ca	39 - 106	41 - 207	-	-	99 - 543	86 - 404
Mg	4.6	0.15 - 127	-	-	1,474 - 3,247	86 - 1,114
Ca Hd.	75 - 246	130 - 250	-	-	130 - 2,800	150 - 1,000
SO_4	73 - 265	149 - 7,280	2,300 - 3,300	1,840 - 2,300	6,000 - 80,100	7,000 - 27,000
Cl	3.0 - 3.2	3.3 - 5.1	3.30	2.8	19 - 61	5.5 - 12.0
Tot. Hd.	-	220 - 750	1,100	707	3,600 - 15,000	1,400 - 6,250
Alk.	46 - 130	<0.05	<1.0	<1.0	<0.05	<0.05
pH	7.7	2.95-3.35	2.6 - 3.8	2.4 - 5.1	2.10-2.70	2.4 - 3.2
Acidity	8 - 22	1,786 - 6,248	310 - 2,700	53 - 2,970	10,000 - 98,000	1,450 - 22,000
Fe	0.12- 0.32	344 - 2,220	11 - 591	20 - 473	4,472 - 15,200	1,500 - 3,800
Al	<0.01	128 - 445	38 - 244	5.2 - 138	6,141 - 11,920	562 - 3,247
Cu	<0.01	1.7 - 3.0	2.0	0.95	64 - 129	7.3 - 28
Zn	0.02	3.7 - 17	0.6 - 5.2	0.3 - 5.2	207 - 335	24 - 66
Co	<.01	<0.01 - 1.06	<.01	<.01	43 - 105	2.3 - 14.2
Ni	0.03-0.12	3 - 10	4.1	2.3	24 - 188	15 - 53
As	<0.05	<0.05 - 0.01	<0.05	<0.05	.94 - 9.0	.11 - 2.0
TDS	199 - 404	342 - 5,572	614 - 4,600	220 - 3,645	12,870 - 94,852	3,757 - 21,400
TOC	1.0	11.0	3.20	-	460	30 - 33
NH4	<0.05	4.50	0.5	3.5	5.0	1.2 - 3.3
D.O.	2.8	0.2	-	-	0.15 - 0.50	1.2 - 1.4

Water quality in disturbed bedrock (represented by ME-83-1) located 30 feet from Delta Taxiway (Figure 1) illustrates significant higher concentrations of acidity, sulphate, calcium, magnesium and dissolved metals compared with background levels. The lower values were measured shortly after bedrock was blasted with steady increases measured since then. Sodium and chloride concentrations and metals of lead, antimony, selenium and tin are within background concentrations. The pH is within the range of 2.95-3.35.

The water quality discharge from the taxiway underdrain (Station A, Figure 1) and storm water interceptor (Station C) reflect acid drainage from disturbed bedrock around the taxiway. Again, the concentrations of acidity and metals show increases since excavation began.

The water quality within the waste rock pile indicate extremely acidic conditions. The values in Table 4 are those recorded for MW #3 and ME-83-5, which represent the highest acidic levels of the monitors placed in the waste rock.

The pH is in the range of 2.1-2.7 and acidity values of 10,000-98,000 mg/L are reported. Iron and aluminum are at saturation levels.

The seepage of waste rock ground water to the holding pond indicates a substantial reduction in concentration from the high strength acid water found inside the pile. The sulphate, acidity and metal concentrations are reduced by 30-50 percent. The pH, however, remains low.

Dissolved oxygen (DO) measurements of 2.8 mg/L were noted in background water, whereas in areas of disturbed bedrock and monitors placed in the waste rock pile, values of less than 1 mg/L were reported. This suggests the presence of aerobic biological activity which is supported by the high values of total organic carbon (TOC) and ammonia present in the waste rock pile and fractured bedrock. *Thiobacillus ferroxidans* was positively identified in MW #1 just below the water table at depths of 25 to 27 feet below ground surface.

DISCUSSION

Acid drainage is produced from the oxidation of metallic sulphide minerals, particularly those containing iron. Disturbance of bedrock by blasting and excavation opens many new fractures, particularly along cleavage planes in the mineralized slate. Once these fresh surfaces are exposed to oxygen and water, the result is a release of acid drainage (low pH water) and heavy metals to surface and ground water systems of the immediate and downstream area.

In theory, pyrite can oxidize chemically and biologically to produce acid drainage. Research has demonstrated that direct chemical abiotic oxidation of pyrite is too slow to generate the amount of acidity observed in nature. The bacterium *Thiobacillus ferroxidans* is always present in acid mine waters and there is considerable evidence which suggests that the biological mechanism is significant, acting as a necessary catalyst for the relatively more rapid ferric iron (Fe^{3+}) chemical oxidation of pyrite (Singer and Stumm, 1969). The presence of this bacterium can increase the rate of oxidation 10^5 to 10^6 times the inorganic chemical reactions (Lacey and Lawson, 1979).

Thiobacillus ferroxidans in a unique bacterium which obtains its energy for growth from the oxidation of reduced sulphur compounds (sulphides) and ferrous iron. There are other species of *Thiobacillus* which are capable of producing small quantities of acid drainage, however, none are as significant as *Thiobacillus ferroxidans* (McCready, 1983). The acid generation process is illustrated as follows:

$$\underset{\text{pyrite}}{4\ Fe\ S_2} + \underset{\text{oxygen}}{15\ O_2} + \underset{\text{water}}{14\ H_2O} \longrightarrow \underset{\substack{\text{ferric/iron}\\\text{hydroxide}}}{4\ Fe\ (OH)_3} + \underset{\substack{\text{sulphuric}\\\text{acid}}}{8H_2\ SO_4}$$

Although the detailed pathway describing the function of Thiobacillus ferroxidans is complex and subject to debate, the major reactions described in Table 5 reflect current understanding of acidification mechanisms. The actual reaction process is believed to occur in a multi-stage sequence dependent upon the activity of Thiobacillus ferroxidans and pH (Kleinmann et. al., 1981). During the first stage of this process fine grained pyrite is oxidized either by Thiobacillus ferroxidans or by air with equal amounts of acidity produced by the oxidation of sulphide to sulphate (Reaction 1) and by the hydrolysis of ferrous iron (Fe^{2+}) (Reaction 2). The fall of pH is moderated as it approaches 4.5 by a pronounced decrease in the rate of reaction 2 (Stumm an Morgan, 1970).

As abiotic oxidation of Fe^{2+} slows, bacterial oxidation accelerates and takes over the primary role of oxidizing Fe^{2+}, thereby allowing reaction 2 to continue producing acidity and ferric hydroxide. This initiates Stage 2 of the reaction which rapidly depresses the pH and acid production is accelerated.

At pH less than 3, the increased solubility of iron and the decreased rate of $Fe(OH)_3$ Precipitation results in increased ferric iron (Fe^{3+}) activity which is the major oxidant of pyrite. Stage 3 begins as Fe^{3+} activity becomes significant at a pH of approximately 2.5. The model involves the oxidation of Fe^{2+} to Fe^{3+}, which consumes 1 mole of acidity, followed immediately by the reduction of Fe^{3+} by pyrite which generates more Fe^{2+} and 16 moles of acidity. Once in the third stage, the acid generation processes are extremely difficult to arrest. The rate of determining step is the oxidation of Fe^{2+}, which is governed by the rate of bacterial oxidation.

TABLE 5

1. $FeS_2 + 7/2\ O_2 + H_2O \longrightarrow Fe^{2+} + 2SO_4{}^{2-} + 2H^+$
2. $Fe^{2+} + 5/2\ H_2O + 1/4\ O_2 \longrightarrow Fe(OH)_3 + 2H^+$
3. $Fe^{2+} + 1/2\ O_2 + H^+ \longrightarrow Fe^{3+} + 1/2\ H_2O$
4. $FeS_2 + 14Fe^{3+} + 8H_2O \longrightarrow 15Fe^{2+} + 2SO_4{}^{2-} + 16H^+$

STAGE 1

Reaction 1 proceeds both abiotically and by direct bacterial oxidation

Reaction 2 proceeds abiotically, slows down as pH falls

Chemistry: pH above approximately 4.5, high sulfate, low iron, little or no acidity

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STAGE 2

Reaction 1 proceeds both abiotically and by direct bacterial oxidation

Reaction 2 proceeds at rate determined primarily by activity of _T. ferroxidans_

Chemistry: approximate pH range of 2.5-4.5, high sulfate, acidity, and total iron increasing, low Fe^{3+}/Fe^{2+} ratio

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STAGE 3

Reaction 3 proceeds at a rate totally determined by activity of _T. ferroxidans_

Reaction 2 proceeds at a rate primarily determined by rate of reaction 3

Chemistry: pH below approximately 2.5, high sulfate, acidity, total iron and Fe^{3+}/Fe^{2+} ratio

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after Kleinmann _et_. _al_. 1981.

The reaction process just described indicates that without the presence of the bacterium Thiobacillus, the reaction is unlikely to proceed below a pH of 4.5. As this bacteria has been identified in the waste rock pile, it is postulated that bacterial oxidation of ferrous iron is ongoing within the oxygenated vadose zone and capillary fringe of the fractured bedrock adjacent the taxiways and runways and in the waste rock pile. The bacterium and ferric iron ions move vertically with ground water flux to the lower zones in the bedrock and waste rock pile. Contrary to earlier beliefs that the bacterium requires a solid medium to grow on, it has been recently demonstrated that these organisms are capable of surviving in mobile supernatant waters (Ferguson, 1985). In the lower anaerobic zones, pyrite dissolution is mostly chemical (ferric oxidation), resulting in large quantities of acid drainage production. This is contrary to earlier beliefs that an ample supply of oxygen is required. Oxygen is required only initially to regenerate Fe^{3+} by aerobic bacterial activity. Fe^{3+} is itself the specific pyritic oxidant and oxidizes pyrite in the absence of bacteria and oxygen (Baker and Wilshire, 1979). For the initial stages of the reaction, the highly permeable nature of the protective cap over the waste rock and the absence of a confining vegetative mat over the fractured bedrock provides an accessible pathway for oxygenated rainwater to enter the waste rock and bedrock.

There is also ample supply of nutrients for bacteria growth supplied by urea application and eroding soil conditions. The bacterium requires water, oxygen, carbon dioxide and ammonia as well as small amounts of phosphorus, calcium and magnesium (Duncan, 1972), all of which are identified in the waste rock and ground water (Table 4). As the oxygen levels decrease in the lower regions of the waste rock pile, oxygen can be obtained from organic carbon, which is present in elevated concentrations, and thereby sustain the minimal bacterial growth required in the later stages of the reaction.

It is evident from the reaction processes that if bacterial oxidation is inhibited, the acid production would cease at a pH of 4.5 due to the stop in abiotic oxidation of Fe^{2+} as described in Reaction 2. At pH values in excess of 4.5, ferric iron is in the form of the hydroxide and is not available to oxidize pyrite as illustrated in Reaction 4. As pH drops, it is much more difficult to return an acid producing system to Stage 1 (Kleinmann *et*. *al*., 1981).

The steady state activity is determined by the combined effects of bacterial oxidation of Fe^{2+}, reduction of Fe^{3+} by pyrite and the formation of associated sulphate and hydroxyl complexes. Unfortunately, the system is never in equilibrium, being controlled by kinetic processes which cannot be described in terms of equilibrium chemistry (Kleinman *et*. *al*., 1981). Any one reaction constantly buffers another reaction. For instance, equilibrium chemistry dictates the pH of the discharge waters should be lower than that reported for such high acidity in the waste rock pile, yet a pH greater than 2.2 is maintained. This is due to release of hydroxyl ions (OH^-) as the iron solubility dramatically increases below pH 3 and iron hydroxides resolubilize. Other reactions such as chemical reductions of nitrate and sulphate to nitrogen and hydrogen sulphide in addition to anionic exchange or nutrient uptake of nitrates tends to buffer acidity whereas oxidation of ammonia, cationic exchange of calcium or nutrient uptake adds acidity (Dickson, 1980).

The mechanisms just described explain the high strength of acid drainage generated in the underdrain systems and waste rock pile which was not anticipated. It also explains the observed lag time of approximately 3-4 months before generation of the acid drainage at the Airport.

The four years of water quality data collected indicate progressive increases in strengths of acid water over time within the waste rock pile. It appears that a peak was reached in acid formation some time during early 1986, with acidity levels in the late fall of 1986 only 25 percent of that reported for 1984-1985. Concentrations of metals and sulphate ions have also shown decreased levels in 1986 in monitors in the waste and the two seepages at the toe of the rock pile. It is inevitable that acid production should slow when the metal availability becomes exhausted within the waste rock pile. As stated earlier, waste samples collected in early 1984 indicated a possibility that 30-60 percent of the metals may have already been leached to that time (a period of two years since acid inception). Further water quality monitoring is required for confirmation.

Although the seepage at the base of the waste rock pile illustrates decreases (in 1986) of sulphate, acidity and metals, there is no associated decrease of calcium, magnesium and aluminum. This may reflect dissolution of the glacial till by the acid water. This dissolution process is well documented in the literature (Fuller, 1981).

Acid drainage concentrations are continuously increasing (including steady increases through 1986) in the bedrock beneath the waste rock (evidenced by water quality in ME's-83-3(2), 4(2) and 8(2)), indicating movement with the ground water flux. The reddish brown sludge observed in the swamp north of the settling ponds may be metal hydroxide precipitates from ground water discharge of acid waters originating from the waste rock in addition to resolubilized metal hydroxide sludge from the base of the collector and settling ponds. This is supported by high vertical hydraulic gradients measured in the area. Vertical gradients of 0.3 to 0.8 and horizontal gradients of 0.01 and 0.02 have been measured in the waste rock (Lund, 1985). Slug tests conducted by Monenco (1983) report hydraulic conductivities of 6.67 x

10^{-4} cm/sec for the bedrock and 2.55 x 10^{-4} cm/sec for the waste rock. This indicates a rate of movement approximately 2 feet per day through the waste rock (Lund, 1985). Head of water in the settling ponds is about 6-8 feet higher than ground surface in the adjacent swampy area, indicating a strong tendency for movement out the bottom of the ponds. Figure 2 illustrates the resultant ground water flow pattern at the site.

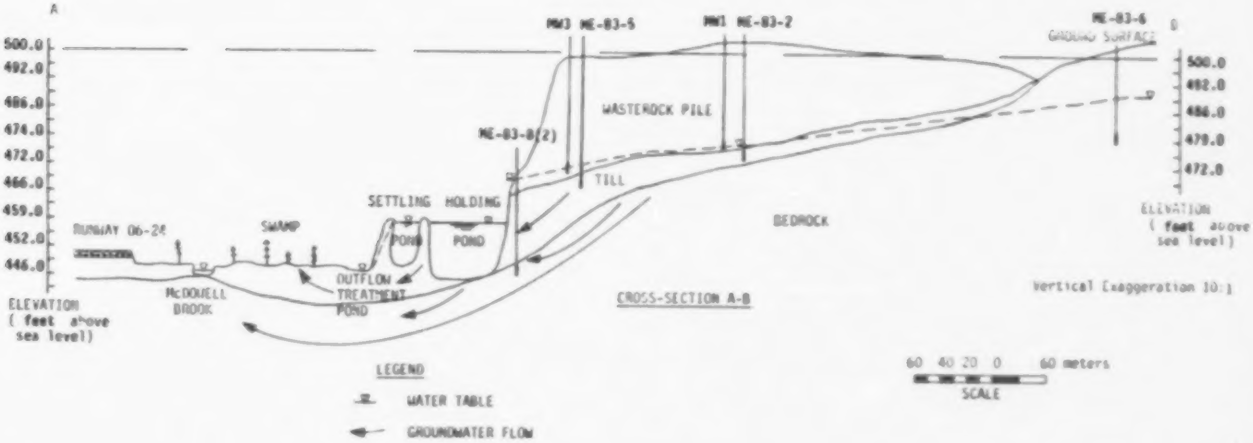
A
500.0
492.0
486.0
474.0
472.0
466.0
459.0
452.0
446.0
ELEVATION
(feet above
sea level)
RUNWAY 06-24
SWAMP
McDOWELL
BROOK
SETTLING
POND
HOLDING
POND
OUTFLOW
TREATMENT
POND
ME-83-8(2)
MW3
ME-83-5
MW1
ME-83-2
ME-83-6
B
GROUND SURFACE
WASTEROCK PILE
TILL
BEDROCK
500.0
492.0
486.0
479.0
472.0
ELEVATION
(feet above
sea level)
Vertical Exaggeration 10:1
CROSS-SECTION A-B
LEGEND
WATER TABLE
GROUNDWATER FLOW
60 40 20 0 60 meters
SCALE

Figure 2

Water quality in the fractured bedrock adjacent the Delta Taxiway also showed a progressive increase in acid strength over the period of record. The distance upgradient (from the blast area) at which acid drainage began forming increased progressively outward over time. The first signs of acid drainage in ME-83-6 appeared in October 1984, a distance of approximately 250 feet from the taxiway. This is a significant consequence in view of future blasting impacts in similar slate terrain. The data for ME-83-1 and ME-83-6 indicate an improvement in water quality in 1986 compared with data for earlier years. There is, however, no evidence of a downward trend in acid concentrations. Continued monitoring is required to determine when downward trends became apparent.

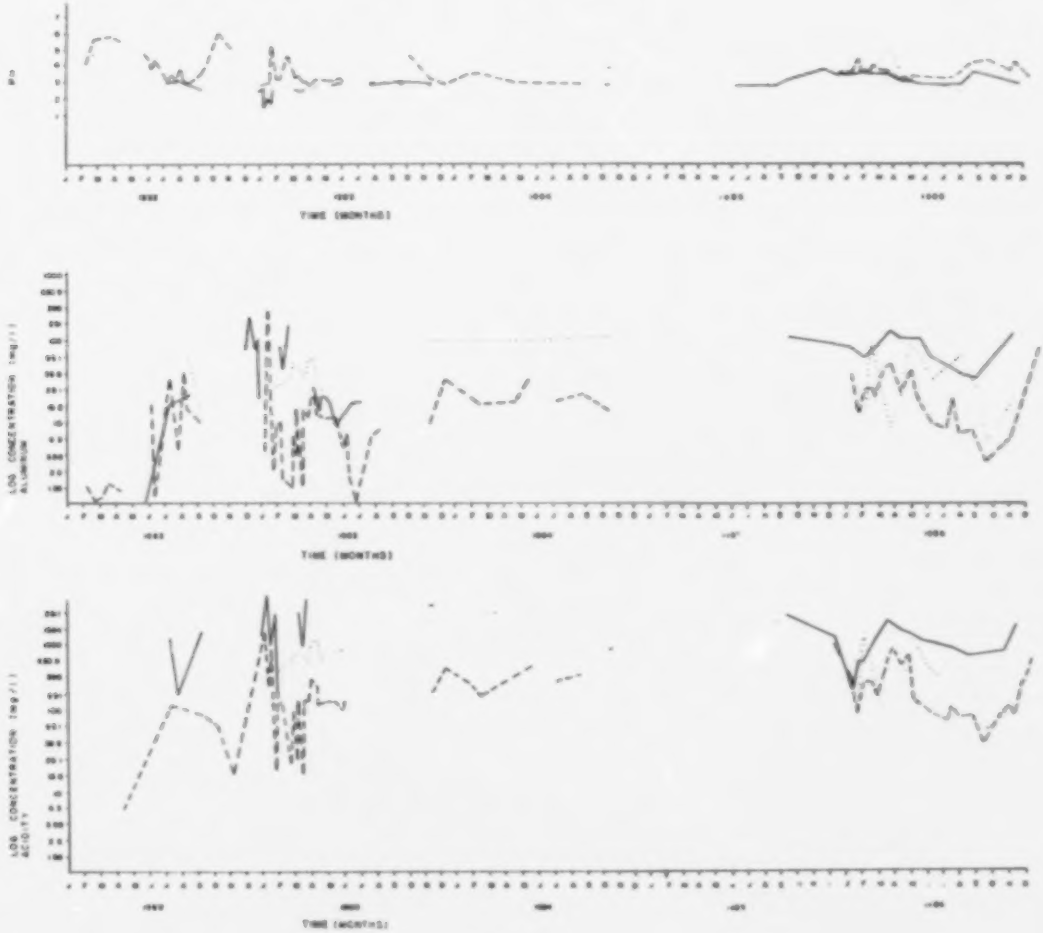

Figure 3

McDowell Brook receives treated acid water and untreated acid water which bypasses the treatment plant through the storm water interceptor (Station C), ground water leakage through the base of the waste rock and sludge lagoons and storm event overloads. The data illustrates there is no improvement of water quality in 1986.

Although the data is highly variable, a number of trends are apparent in Figure 3. The acidity and metal concentrations at the McDowell Brook station are consistently lower and pH higher than reported values for the underdrains and storm water collector. This indicates the treatment plant is effective in removing quantities of acid load from the downstream watercourses. The data, however, shows no improvement in treatment effectiveness over previous years. Although it appears that acid loads from the waste rock have decreased as a function of metal exhaustion, a subsequent decrease is not apparent in the headwaters of McDowell Brook. As well, there is no apparent improvement in the quality of water from the discharge of the underdrains and storm water interceptor, despite lower reported values of dissolved metals in the fractured bedrock monitors ME-83-1 and 6 for the period 1986.

The data in Figure 3 indicates increased acidity and metal concentrations during spring snowmelt and fall rains in the upper underdrains, a similar trend noted for the upper

McDowell Brook station. The surface water interceptor does not exhibit this trend although the acidity and aluminum concentrations are consistently lower at this station than in the underdrain discharge. The pH is closely coincident during each period of measurement in the upper underdrain and storm water collector. Such similar pH with high variance of acidity and aluminum illustrates the nature of non-steady state conditions discussed earlier.

A cursory comparison was made of the reported values for the storm water interceptor and upper underdrains and daily precipitation recorded at the Airport. The analysis reveals that the high values of acidity and metals occur after high intensity rainfalls in the spring and fall for the upper underdrain system with lower values reported during dry periods and periods of snow cover. This is likely related to flush out of stored acid drainage. Kleinmann *et*. *al*. (1981) note that acid drainage remains elevated, for three to four days following a rainfall event as a result of increased dissolved oxygen levels and subsequent increase in biological activity.

The surface water interceptor exhibits higher values of acidity and aluminum during dry cold spells in the winter and during low intensity, long duration rainfall events during the rest of the year. The lowest values are reported during high intensity rainfall events, resulting from dilution of non-acidic surface waters.

Computer modelling techniques are being applied to map the ground water chemical flux in and around the waste rock pile and fractured bedrock in order to further define the pathways of the acid drainage generation at the Airport. A more detailed statistical analysis is also planned to determine the magnitude of storm related acid loads in relation to climatic factors of storm intensity and duration. Continued monitoring is ongoing by Transport Canada to determine the time related magnitude of the acid drainage problem at the Halifax Airport and to measure improvements in water quality resulting from current mitigation programs, such as the re-capping of the waste rock pile and vegetative program to reduce infiltrating oxygenated water. Recently, monitors have been installed alongside the Delta taxiway and water samples are collected monthly.

ACKNOWLEDGEMENTS

The author gratefully acknowledges the assistance of Dan Thirumurthi (TUNS), John Machell (Environment Canada) and Terry Hennigar (Environment Canada) for technical guidance, and also to the Centre for Water Resources Studies at the Technical University of Nova Scotia for financial support and enabling the research project to continue. Gratitude is extended to Ross McCurdy and Debbie Borgal of the Nova Scotia Pathological Institute for providing laboratory services in the chemical analyses of the water samples. I also wish to thank Transport Canada for providing the site for study and to Larry Butler, Eloi Gaudet and Paul Goudy for providing technical assistance in gathering field information and additional water quality data.

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SURFACE AND GROUNDWATER IMPACTS OF ACID MINE DRAINAGE FROM THE MEGUMA SLATES OF NOVA SCOTIA

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Presented to the Acid Mine Drainage
Seminar, Halifax, Nova Scotia
March 26-27, 1987

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SURFACE AND GROUNDWATER IMPACTS FROM ACID MINE DRAINAGE OF THE MEGUMA SLATES

SUMMARY

In Nova Scotia a number of geologic units occur with sulphide mineralization. These rocks range from those associated with coal seams of upper Carboniferous age, to various types of sulphide mineralization in the Ordovician metasediments of the Meguma Group. The former units occur in Northern, Central, and Eastern Nova Scotia. The latter group underlies the southern half of the mainland portion of the Province. The interest in these sulphide mineralized zones from a water resources perspective relates to the potential for acid generation and metal liberation. Another concern relates to the quality of groundwater from wells penetrating aquifers in the Halifax Formation, the upper and more highly mineralized component of the Meguma Group.

Documented cases of fish kills, water supply closures, and water quality problems at a fish hatchery have led to widespread concerns related to water impacts of acid mine drainage from these slates. A wide variety of pertubations have resulted in acidity production and metal releases from the materials exposed in these areas. The problem is aggravated by the action of Thiobacillus ferroxidans.

Work initiated in 1985, with follow up in 1986 included mapping existing slate quarries in southwestern Nova Scotia. A reconnaissance survey in Lunenburg and Queens Counties identified 110 potential Meguma slate quarries from aereal photographs, and other sources of mapping. Field truthing of these sites revealed that fifty (50) of these quarries had in fact exposed sulphide mineralized portions of the Halifax Formation. Water quality sampling and analyses at points of inflow to, standing water in, and outflow from the quarries was carried out. pH of these waters varied from 2.7 to 6.4 in a total of 135 samples where 75% of the samples had a pH of less than 4.0.

The water management implications of acid mine drainage from these rock units are not well understood or recognized. Much work is required to evaluate the impacts of the acid drainage and metal liberation phenomena on the water resources. A current effort to this end is a study to identify acid drainage risks, formulate prediction models of occurrence, measure the degree of impact, and develop the criteria for mitigation strategies.

1.0 INTRODUCTION

Now a well recognized phenonemon, the first known reference to acid drainage in North America (according to Hawley, 1972) was in 1698:

> "I have reason to believe that there are good coals (here) also for I have observed (that) the runs of water have the same color as that which proceeds from the coal mines in Wales."

Although commonly associated with coal mining, where it has been most extensively studied, acid drainage is now recognized in conjunction with all types of metallic sulphide mineralization.

Acid production occurs when sulphides are exposed to water and oxygen. Although the action can, theoretically proceed chemically, the rate of reaction is greatly accelerated by bacterial activity. Work by Leathen et al. (1956) illustrates this. About 200 mg/l of ferric iron was obtained from ferrous iron in mine drainage water in three days through oxidation by Thiobacillus ferrooxidans. Purely chemical oxidation required more than two years to obtain the same effect.

In Nova Scotia there are a number of geoligical units with significant sulphide mineralization varying widely in age and rock type from the older Cambro/Ordovician Meguma metasediments to the younger Upper Carboniferous coals. The Carboniferous rocks occur in the northern half of the province while the Meguma metasediments underlie the southern half of mainland Nova Scotia (Figure 1).

Several sources, or potential sources, of acid drainage have been documented in the Cape Breton coal fields by the Environmental Protection Service (McLeod, 1984). Included in these sources are such things as coal preparation facilities, raw coal and coarse tailings stock-piles, stone tips, acid drainage from bootleg pits and improperly abandoned overflowing coal exploratory drill holes.

The Meguma Group and its associated sulphide mineralization, however, may pose a greater threat to surface and groundwater sources than the coal areas. The focus of this paper is on the Meguma Group and the associated concerns with respect to acid mine drainage.

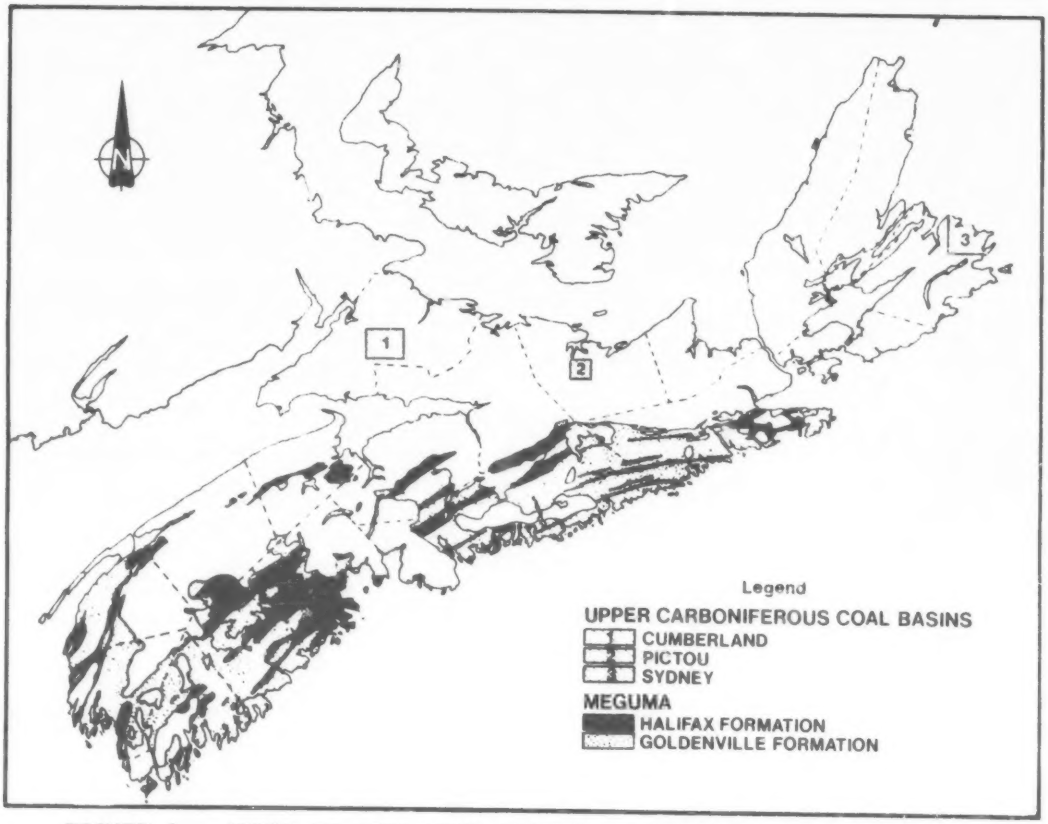

N
1
2
3
Legend
UPPER CARBONIFEROUS COAL BASINS
1 CUMBERLAND
2 PICTOU
3 SYDNEY
MEGUMA
HALIFAX FORMATION
GOLDENVILLE FORMATION

FIGURE 1 - AREAS OF ACID MINE DRAINAGE POTENTIAL IN NOVA SCOTIA
(Modified from Cooke, 1972)

2.0 THE MEGUMA GROUP

The Meguma Group is generally considered to be composed of two units: the Goldenville Formation, older of the two, consisting of thickly bedded metagreywacke with thinly interbedded slate and slate metawacke; and the Halifax Formation consisting of grey and black slates with thinly interbedded pyritiferous metasiltstone and sandstone (Porter-Dillon, 1986).

Three mapable units have been recognized in the Halifax Formation. The Moshers Island Member, which conformably overlies the Goldenville Formation is a green to grey-green parallel laminated argillite. This unit is typically manganiferous, carbonate bearing and often contains sulphide mineralization. Overlying this unit is the Cunard Member, consisting of black slate interbedded with thinly bedded pyritiferous metasiltstone. The Felzen Member overlies the Cunard and consists of grey slates with interbedded bioturbated sandstone beds.

Considerable work has been done on the Meguma Group in recent years by Zentilli et al. (1984, 1986a, 1986b), Schenk (1970), O'Brien (1986), and others. Because of its fine grained nature, structural complexity, and generally higher degree of sulphide mineralization, the Halifax Formation is the rock unit of most concern with respect to acid drainage.

3.0 THE PROBLEM IN GENERAL

If sulphide mineralized rocks are left undisturbed, acid drainage from these units is not a concern. However, various types of human activities, such as construction and mining expose sulphides to oxygen and water. Further crushing through ore benefication and aggregate production activities greatly increases the exposure by increasing surface area.

It is often difficult or impracticable to avoid development activities in the Halifax Formation, mainly because the rock unit is geographically so widespread in mainland Nova Scotia and is either exposed as bedrock outcrop or is covered with a thin veneer of overburden. Highway construction, quarrying, industrial and urban development is going on in many of the areas underlain by this Formation, where it is being exposed and/or redistributed for various purposes.

The Halifax Formation is widely used as a cheap and convenient source of aggregate. It is easily mined, often requiring little or no blasting, and being so widespread, is often very close to development activities and demand areas. Much of the slate aggregate is used as road fill and resurfacing material in Nova Scotia. Additionally, the material is often used to surface driveways, parking lots and used as fill for building sites.

The general impacts of acid run-off have been widely documented in Nova Scotia. Acid drainage has caused fish kills, the closure of at least one fish hatchery, disruption of public water supplies, some contamination of private wells, and damage to engineering works.

4.0 SURFACE WATER IMPACTS

The problem first came to prominence with the construction of the Halifax International Airport in the headwaters of the Shubenacadie River and Salmon River Systems. A major fish kill and fish hatchery closure on the Shubenacadie River system was attributed by Scott (1961) to construction activities in, and disturbance of, the underlying slates. It was not until 1976, when a new hangar was constructed in slate, that intensive study of the problem was initiated. At this time the Environmental Protection Service (1976) linked fish kills to perturbations of slate at the airport (Table 1). Porter-Dillon (1985) suggests that the poor health of fish in the Salmon River system can be attributed to acid drainage originating at the airport. Sampling of the runoff from the slate stock pile indicated very high values of metals and acidity. Examples of high values found include Fe at 5030 mg/l, Al at 2330 mg/l, SO_4 at 30,900 mg/l, and acidity as high as 27,400 mg/l as $Ca\ CO_3$ (Lund, 1985).

Slate used to resurface a road at Union Square in Lunenburg County is believed responsible for a major fish kill in August, 1977 (Pettipas, 1979; Thompson, 1978). The quarry from which the slate was obtained is a known occurrence of sulphide mineralization (King, 1985).

Contamination of the Shubenacadie River System has caused occasional disruption of public water supplies of the communities of Elmsdale, Lantz, and Enfield (Thompson, 1978). Other communities using the River below Lantz are probably less susceptible to acid drainage as the river flows through areas of Carboniferous rocks, including limestones and gypsum, and also benefit from the dilution factors of increased natural river flows.

Ogden and Machell (1985) studied the ionic and mass balances in Soldier Lake, an undeveloped, oligotrophic, brown water lake near the Halifax International Airport. This lake has been acidified from leaching of pyritized metalliferous slates since the 1960's as a result of airport related activities. The water impacts identified in this study are highlighted by the estimate that over 60% of the H^+ loading to the lake came from only 1.7% of the catchment area. From a water resources protection point of view, control measures implemented in a small segment of such a watershed could significantly reduce acid

TABLE 1. FISH KILL HISTORY IN THE SHUBENACADIE RIVER

(From EPS, 1976)

MAJOR CONSTRUCTION		MAJOR FISH KILLS
Terminal and Runways	1957-1960	September, 1960
Imperial Oil	1959-1960	
IMP Hangar (large)	1959	
IMP Hangar (small)	1961	
Air Canada Hangar	1961	
Air Halifax Hangar	1961	October, 1961
Highways slate needs	1965	October, 1965
Avis Service Station	1966	September, 1966
Highways slate needs	1968	November, 1968
Halifax Flying Club	1970	
Mobil Oil Hangar	1972	
Highway Overpass	1974	August, 1974
Aircon Tank Farm	1975	October, 1975
EPA Hangar	1976	September, 1976

loadings to the downstream system. The management concerns of this phenomena are that a perturbation of natural earth materials in such a small area may have a significant impact on the downstream water resources.

The distribution of sites with the potential for generating acid drainage and impacting on the water resources of Nova Scotia is also illustrated by the widespread activity related to gold mining in the Province. Mossman (1982) lists sixty (60) gold producing areas scattered across southern mainland N.S. where mining and/or prospecting activity has occurred since 1850 (Figure 2). Associated with the gold is a variety of sulphide mineralization including arsenopyrite, pyrite; galena, chalcopyrite, and pyrrhotite all of which produce and release metal ions into the water column when exposed to acid and oxidizing conditions in the presence of Thiobacillus ferrooxidons.

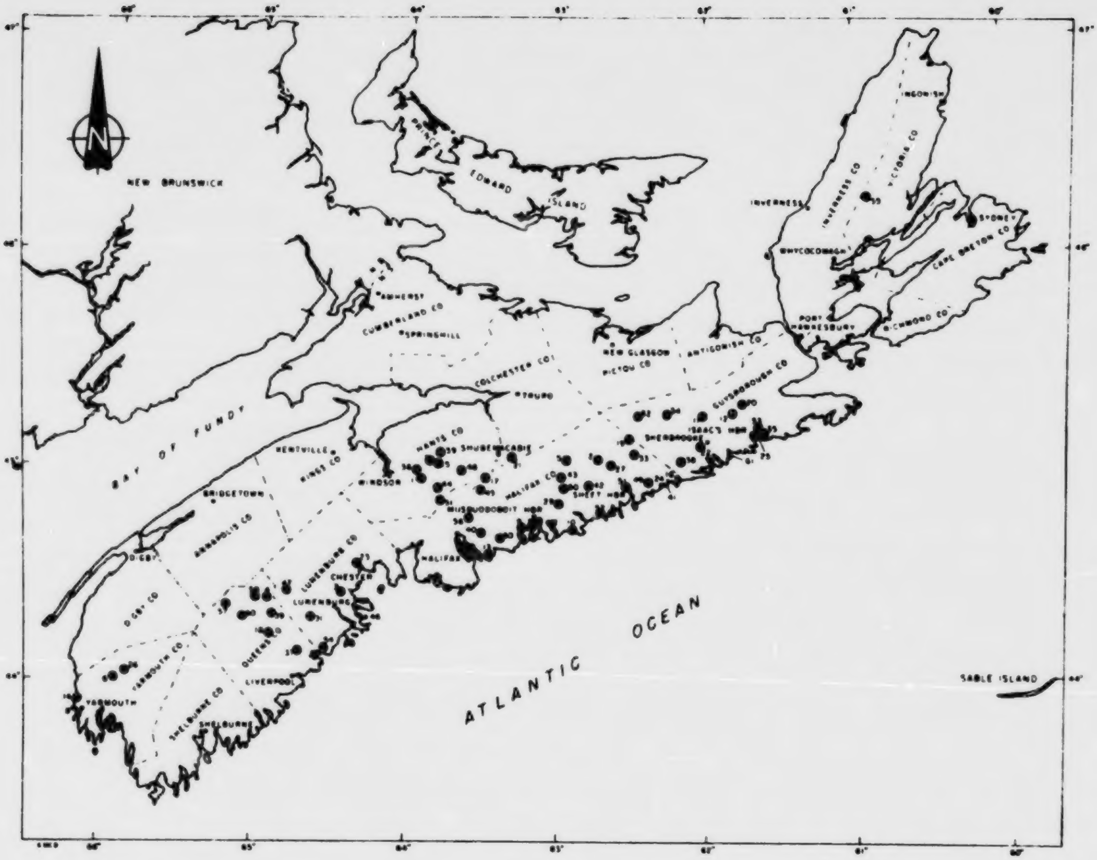

N
NEW BRUNSWICK
PRINCE EDWARD ISLAND
INGONISH
INVERNESS CO
VICTORIA CO
INVERNESS
SYDNEY
CAPE BRETON CO
WHYCOCOMAGH
PORT HAWKESBURY
RICHMOND CO
AMHERST
CUMBERLAND CO
SPRINGHILL
COLCHESTER CO
TRURO
NEW GLASGOW
PICTOU CO
ANTIGONISH CO
GUYSBOROUGH CO
ISAAC'S HBR
SHERBROOKE
SHEET HBR
HALIFAX CO
MUSQUODOBOIT HBR
HANTS CO
SHUBENACADIE
WINDSOR
HALIFAX
KENTVILLE
KINGS CO
BAY OF FUNDY
BRIDGETOWN
ANNAPOLIS CO
DIGBY
DIGBY CO
LUNENBURG CO
CHESTER
LUNENBURG
QUEENS CO
LIVERPOOL
YARMOUTH CO
YARMOUTH
SHELBURNE CO
SHELBURNE
ATLANTIC OCEAN
SABLE ISLAND

FIGURE 2 - GOLD DISTRICTS OF NOVA SCOTIA (after Mossman, 1982)

5.0 GROUNDWATER IMPACTS

The water well contamination by arsenic discovered at Waverley during the mid 1970's was related to gold mining activities, waste rock disposal, and well construction techniques (Gibb et al., 1977). Although these activities were started over 100 years ago the impacts on drinking water supplies are still occurring today and have only recently come to light.

The nature and magnitude of groundwater impacts from acid drainage are not well understood. Contamination of groundwater has been reported in conjunction with coal operations. As an example, O'Bara and Estes (1985) found that springs in the vicinity of, and downgradient from, an abandoned coal strip mine in Tennessee had been contaminated by acid drainage. Typical water quality analyses had pH less than 3, acidities varying from 0 to 118 mg/l, iron up to 259 mg/l and sulphate up to 1241 mg/l.

A few individual well contamination cases from acid drainage are on file in the Nova Scotia Department of the Environment. Typically these have resulted from poor slate management practices or poor well construction, or both. In one case in Fall River, near Halifax, a foundation for a house was blasted in slate, and the excavation subsequently backfilled with the crushed slate and accompanying sulphide minerals. The water supply well, which had insufficient protective casing, was hydraulically connected to the foundation drain, resulting in deteriorated water quality. In another case in Lunenburg County, a cottage driveway was resurfaced with slate containing "fools gold" and other associated minerals. Acid run-off from the driveway reached the area of the dug well and infiltrated the structure through unsealed joints between the well crocks.

One of the better documented cases of groundwater contamination by acid mine drainage in Nova Scotia was investigated by Cross (1976). Roads in Cresthaven Estates, a sub-division in Birch Cove, near Halifax, were being surfaced with slate, some of which was stock piled in a recharge area above a dug well. Cross, as part of a thesis project, had been monitoring this well prior to the incident and had good background data, as illustrated in Table 2. Note the increase in the mean sulphate and the decrease in alkalinity. Although there is no early information on aluminum for this water supply, it is well above background for natural groundwaters in the area. Figure 3 shows a conceptual surface water and groundwater flow system with affects from acid mine drainage.

The problem began about May of 1973 when increased corrosion problems were first noticed. Soda ash (Na_2CO_3) was added to the well at first to raise pH and increase alkalinity. The practice was discontinued because of the large quantity of soda ash required and because the higher pH resulted in the precipitation of aluminum compounds in the hot water tank. The water supply had to be abandoned and for some time a series of cisterns provided the sole water source. Eventually the home was hooked up to the the new municipal water supply from Pockwock Lake.

King (in progress), in a study of acid mine drainage pathways to Bennery Brook, near the Halifax Airport, has found that groundwater does

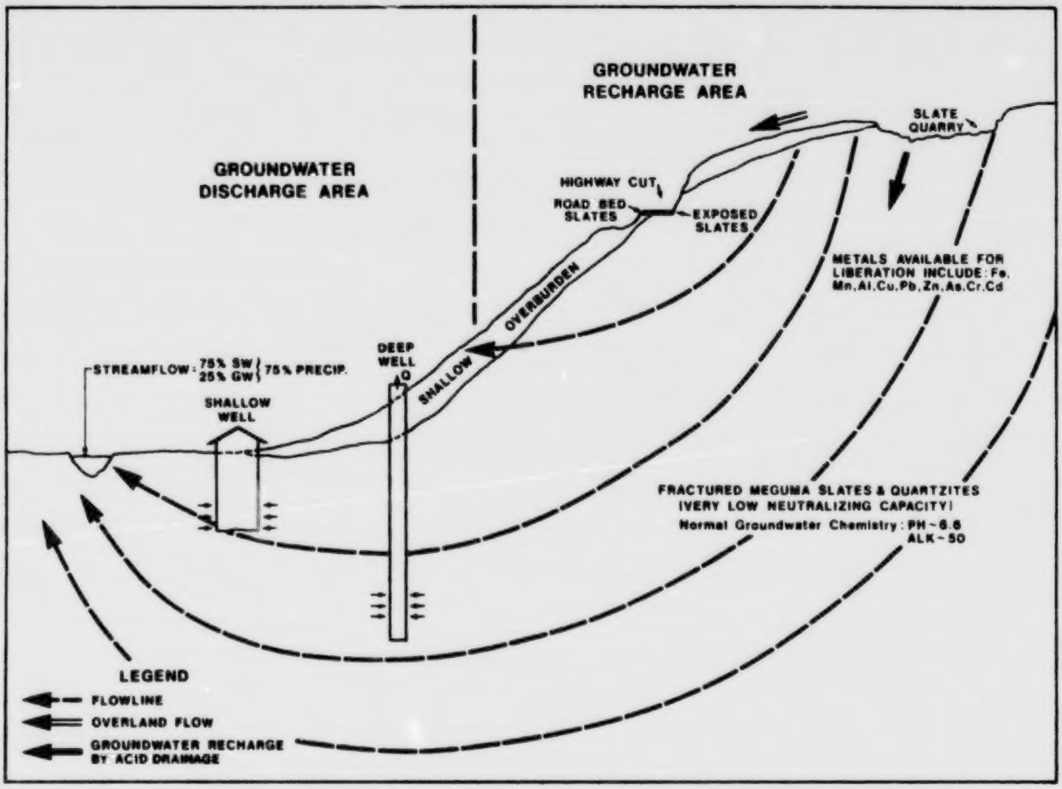

GROUNDWATER
DISCHARGE AREA
GROUNDWATER
RECHARGE AREA
SLATE
QUARRY
HIGHWAY CUT
ROAD BED
SLATES
EXPOSED
SLATES
METALS AVAILABLE FOR
LIBERATION INCLUDE: Fe,
Mn,Al,Cu,Pb,Zn,As,Cr,Cd
OVERBURDEN
SHALLOW
DEEP
WELL
Q
STREAMFLOW: 75% SW
25% GW
75% PRECIP.
SHALLOW
WELL
FRACTURED MEGUMA SLATES & QUARTZITES
(VERY LOW NEUTRALIZING CAPACITY)
Normal Groundwater Chemistry: PH ~ 6.6
ALK ~ 50
LEGEND
FLOWLINE
OVERLAND FLOW
GROUNDWATER RECHARGE
BY ACID DRAINAGE

FIGURE 3 - CONCEPTUALIZED FLOW SYSTEM SHOWING SURFACE AND GROUNDWATER COMPONENTS AFFECTED BY ACID MINE DRAINAGE

provide a transport pathway. Furthermore contaminated groundwater continues to contribute acid and metals to streamflow even during times of prolonged dry periods.

6.0 POTENTIAL IMPACT ON WATER INFRASTRUCTURE

The Public Service Commission of Halifax owns and operates the Pockwock water supply system, the largest (50 migpd capacity) and newest (1977) central system in Nova Scotia. Portions of the pipeline were constructed through areas underlain by the Halifax Formation slates, with no specific provision for protection against the highly acidic conditions that may exist.

TABLE 2. WATER QUALITY CHANGES IN A WELL NEAR CRESTHAVEN ESTATES THROUGH 1972 TO 1976. (After Cross, 1976)

	THESIS SAMPLING AUGUST 1972 - AUGUST 1973 (49 Samples)			N.S. DEPT. OF THE ENVIRONMENT SAMPLING JUNE 1974 - JUNE 1976 (23 Samples)		
PARAMETER mg/l	MIN.	MEAN	MAX.	MIN.	MEAN	MAX.
Na	12.3	38.9	92.5	16.0	55.6	102.0
K	0.35	1.05	2.1	1.4	2.4	3.8
Ca	4.2	7.6	16.6	11.0	27.6	46.0
Mg	0.7	2.0	9.4	7.5	20.5	40
* Alk	7.5	14	25	0	< 1	6
* So_4	4	21.7	82.5	100	275.0	437
Cl	20.0	62.0	145	22.0	73.6	155.0
* Fe	0.005	0.03	0.27	0.1	0.26	2.6
Mn	0.02	0.08	0.26	0.5	1.67	3.6
Cond	96.5	260	525	356	725	957
* pH	5.2	5.8	6.4	4.0	4.5	6.5
* Al	No analysis			12	29	55 (16 analyses)

* Indicators of acid mine drainage.

With the recent recognition of the acid mine drainage problems related to working in the Halifax Formation (i.e. Airport experiences) and a major failure in the Pockwock pipeline in 1985, the Commission initiated a study to determine if deterioration has resulted to the line due to unfavourable and harsh environmental conditions.

The environment in which the Pockwock pipeline has been installed contains features which may accelerate the rate of deterioration of the facility. Some of the unfavourable conditions include:

- low groundwater pH, frequently observed in the presence of Halifax Formation slate bedrock, and in fills originating within that Formation.

- corrosive conditions associated with the introduction of artificial agents such as road salt, into the installation environment.
- areas of exposure to parallel high voltage transmission lines. High voltage alternating current power lines have the ability to induce electrical current in parallel continuous steel structures.

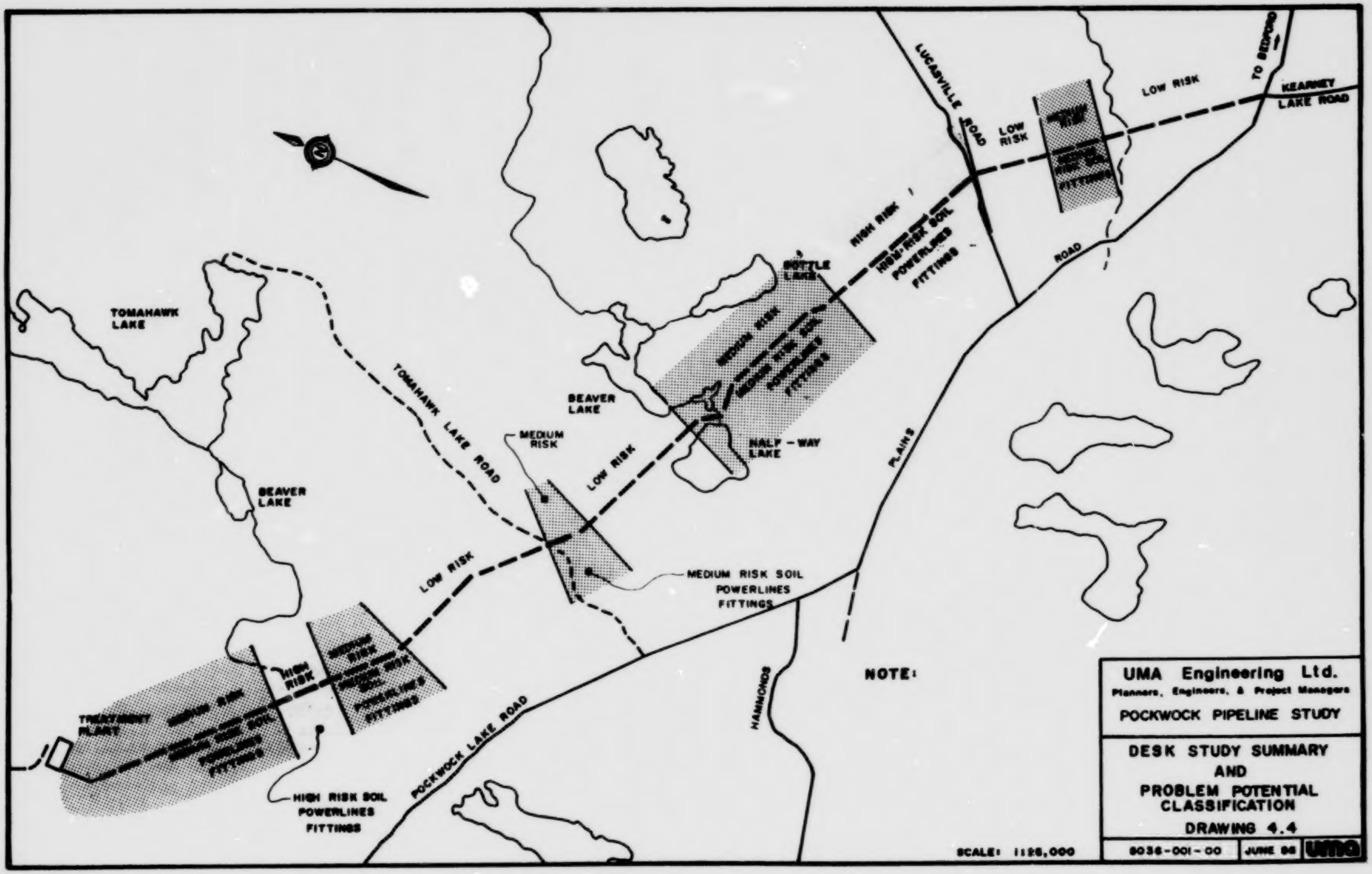
Figure 4 shows the pipeline section from the treatment plant to Kearney Lake Road, Phase I of the study, showing low, medium and high risk areas of concern. These areas were identified and ranked on the basis of design, construction and operational considerations, power line locations, surficial geology analysis, presence of slate bedrock, and drainage conditions. Using this description, and including all factors, field investigations were carried out at 24 test pit locations where samples were collected, piezometers installed for groundwater observations, and various measurements taken for pH.

Among the results of this Phase of the study included:

- pH values of the soil ranging from 4.9 to 7.1.
- pH values of groundwater encountered in the trench ranging from 4.3 to 8.0.
- Acid Producing/Absorbing Potential of the slate as high as 3.06/3.92.
- Electrical half-cell potential values as high as -0.772 volts at joints. Readings more negative than -0.35 volts are indicative of corrosion more than 95% of the time.

Review of the field data failed to identify any major problems with the pipeline, but did result in the decision to carry out additional testing to quantify conditions in certain areas along the pipeline route, and to examine a number of pipeline joints. The pipeline failure in 1985 was found to be related to installation problems at a joint rather than corrosion by acid mine drainage.

However, other information collected during the study indicated that under adverse manufacturing, construction and environmental conditions, pipelines employing steel cylinder lined, prestressed reinforced concrete pipes (one of the types used in the Pockwock line) do fail.


TO BEDFORD
KEARNEY LAKE ROAD
LOW RISK
LOW RISK
LUCASVILLE ROAD
ROAD
HIGH RISK
HIGH RISK SOIL
POWERLINES
FITTINGS
BOTTLE LAKE
MEDIUM RISK
MEDIUM RISK SOIL
POWERLINES
FITTINGS
HALF - WAY LAKE
PLAINS
BEAVER LAKE
LOW RISK
MEDIUM RISK
MEDIUM RISK SOIL
POWERLINES
FITTINGS
TOMAHAWK LAKE ROAD
TOMAHAWK LAKE
BEAVER LAKE
LOW RISK
HIGH RISK
MEDIUM RISK
POWERLINES
FITTINGS
TREATMENT PLANT
HIGH RISK SOIL
POWERLINES
FITTINGS
POCKWOCK LAKE ROAD
HAMMONDS
NOTE:
UMA Engineering Ltd.
Planners, Engineers, & Project Managers
POCKWOCK PIPELINE STUDY
DESK STUDY SUMMARY
AND
PROBLEM POTENTIAL
CLASSIFICATION
DRAWING 4.4
8036-001-00
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uma
SCALE: 1:25,000

FIGURE 4 - SUMMARY OF RISK OF CORROSION OF THE POCKWOCK PIPELINE (UMA, 1986)

As follow-up to this study, it was recommended that (UMA, 1986):

- additional piezometers be installed for monitoring and sampling groundwater in Halifax slate areas.
- a similar study be done south of the Hammonds Plains Road where the pipeline is placed in predominently Halifax slates.

7.0 EVALUATION OF THE MAGNITUDE OF THE PROBLEM

In 1985 the Inland Waters/Lands, Atlantic Region, initiated a survey of slate quarries and repositories in an area of southwestern Nova Scotia (Figure 5), extensively underlain by the Halifax Formation (King 1985, Manchester 1986). The project initially identified 110 potential slate quarries from aerial photographs and available maps. Ground truthing reduced the number of quarries to 50. Each quarry was

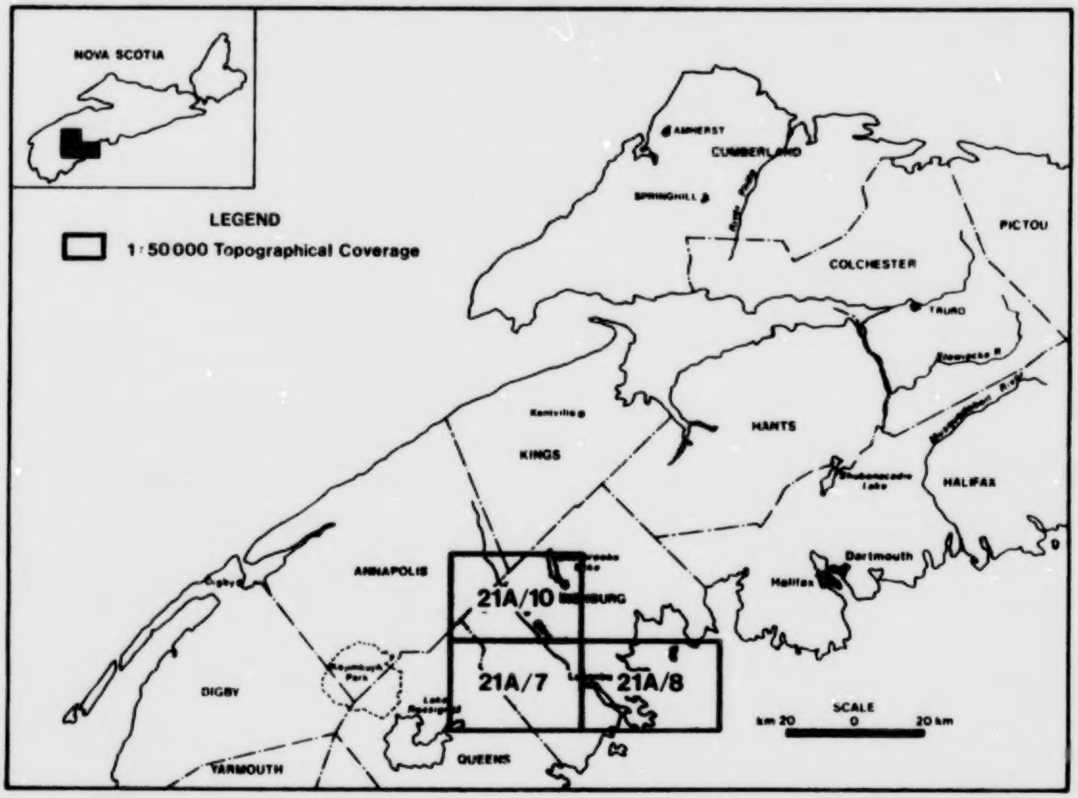
NOVA SCOTIA
LEGEND
1:50000 Topographical Coverage
AMHERST
CUMBERLAND
SPRINGHILL
PICTOU
COLCHESTER
TRURO
HANTS
Kentville
KINGS
Shubenacadie Lake
HALIFAX
ANNAPOLIS
Dartmouth
Halifax
21A/10
21A/7
21A/8
DIGBY
QUEENS
YARMOUTH
SCALE
km 20
0
20 km

FIGURE 5 - LOCATION MAP OF STUDY AREA IN SOUTHWESTERN NOVA SCOTIA (Manchester, 1986)

mapped using the approximate compass and pace technique. Rock samples were taken for later analysis and pH measurements were taken of inflow, outflow and standing waters.

An example of the quarry sketch maps is presented in Figure 6, showing quarry dimensions, other geological features, standing and running water, rock sample and water sample locations. Aerial maps were prepared from base maps (1:50,000 scale watershed maps) available from the Nova Scotia Department of the Environment. In addition to quarry locations, bedrock geology, NAQUADAT water quality stations (Demayo and Hunt, 1975), and hydrometric stations were plotted on the base maps.

The results of this work are summarized in Figure 7. A review of these data show that samples from 43 sites indicate that 75% of the samples had a pH of less than 4.0. Also from these data a bias appears

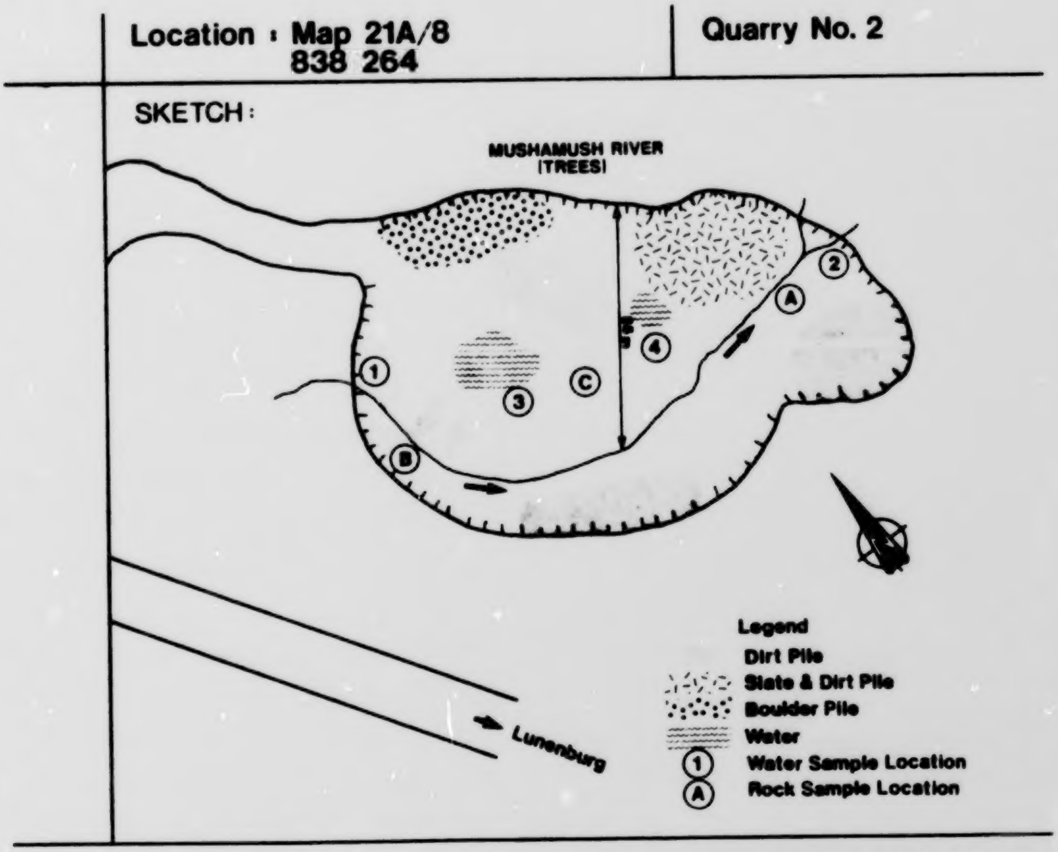
Location : Map 21A/8
838 264
Quarry No. 2
SKETCH:
MUSHAMUSH RIVER
(TREES)
1
2
3
4
A
B
C
Lunenburg
Legend
Dirt Pile
Slate & Dirt Pile
Boulder Pile
Water
Water Sample Location
Rock Sample Location

FIGURE 6 - TYPICAL QUARRY SKETCH UTILIZED IN SOUTHWESTERN NOVA SCOTIA STUDY (after Manchester, 1986)

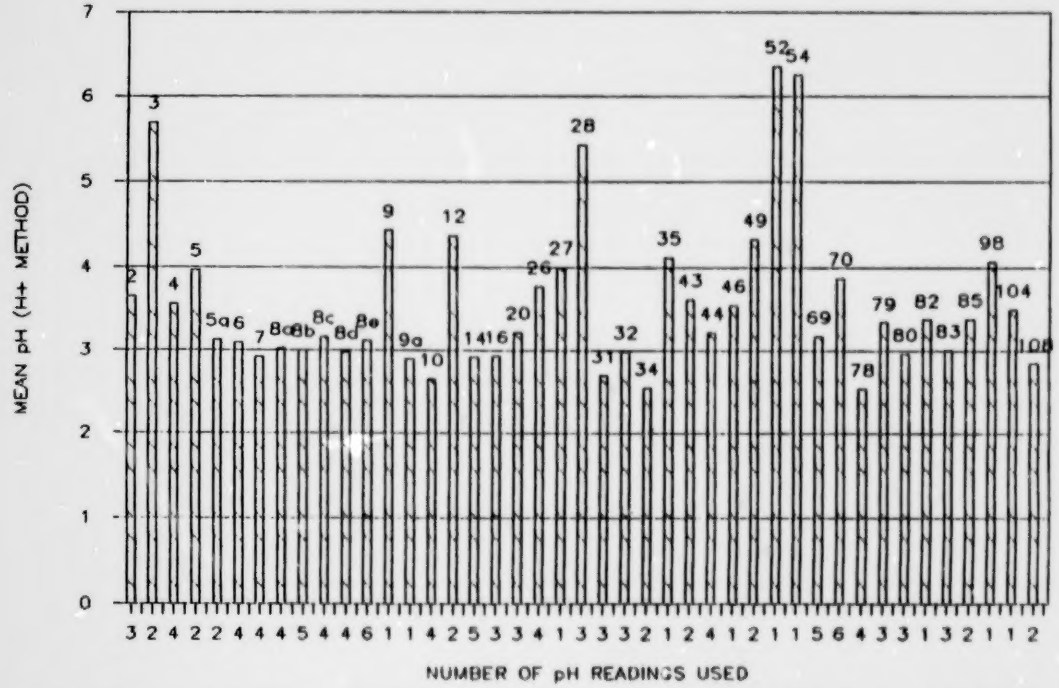

FIGURE 7 – QUARRY pH READINGS IN 1986 - DATA FROM MANCHESTER REPORT (Porter Dillon, 1987)

for less frequently sampled sites, that indicate a higher pH than those sites sampled more frequently. This requires or deserves further study through a more complete sampling program.

Manchester (1986) also found that the pH of standing water in two quarries sampled weekly averaged 3.8 and ranged from 2.7 to 6.4. Flowing waters averaged 4.2 and ranged from 2.8 to 6.4. Results of this weekly pH monitoring in the two quarries showed standing water with a significantly and consistently lower pH than inflow waters. The outflow waters showed much more variability of pH, and generally a lower pH than inflow waters, but consistently higher than standing waters.

Kleinan et al. (1981) found increased acid formation was apparent three to four days after rainfall. Although the data are limited the results of Manchester's work indicate a similar response.

These results lead to the not surprising conclusion that significant acid generation can occur in slate quarries. Although the inventory of slate repositories has yet to be completed, it is apparent that anthropogenic activities in slate are widespread in the study area.

Porter Dillon Ltd., (1987) under contract with the Inland Waters/Lands, Atlantic Region, have produced a preliminary model for assessing acid mine drainage risk (Figure 8) and from this, have prepared a constraint map. The area studied includes Lunenburg, Queens and part of Annapolis County in southwestern Nova Scotia. The initial data base for the model includes available information on bedrock geology, surficial geology, various elements of land use and quarries. The various inputs to the model are:

- lithology
- metamorphic grade of the bedrock
- mineral occurrence
- presence or absence of bedrock imported from "high risk" areas
- buffering capacity of surficial materials
- presence of quarries
- existing/future land use
- site specific watershed characteristics

Not surprisingly, bedrock lithology is the critical factor in the assignment of risk. Additionally, metamorphic grade has been found to be important where the level of risk decreases with increasing grades of metamorphism (the higher the grade the lower the risk). The Halifax Formation, in particular the Cunard Member, presents the highest risk, while the carbonate rocks of the Windsor Group pose no risk. Regional metamorphism progresses from east to west where mineralization grades from chlorite to biotite and on to andalusite/garnet. The bulk effect is that carbonates change to silicate minerals causing increased resistance to weathering and lower permeability, thus decreasing the risk of acid production (Porter Dillon, 1987).

8.0 SUMMARY AND CONCLUSIONS

There are a number of challenges facing water managers when dealing with the acid mine drainage phenomena in the Meguma slates.

As seen from the previous examples and discussion, water related interests in acid mine drainage vary from safety of drinking water to impacts on biota, water supply and infrastructure. The nature of the

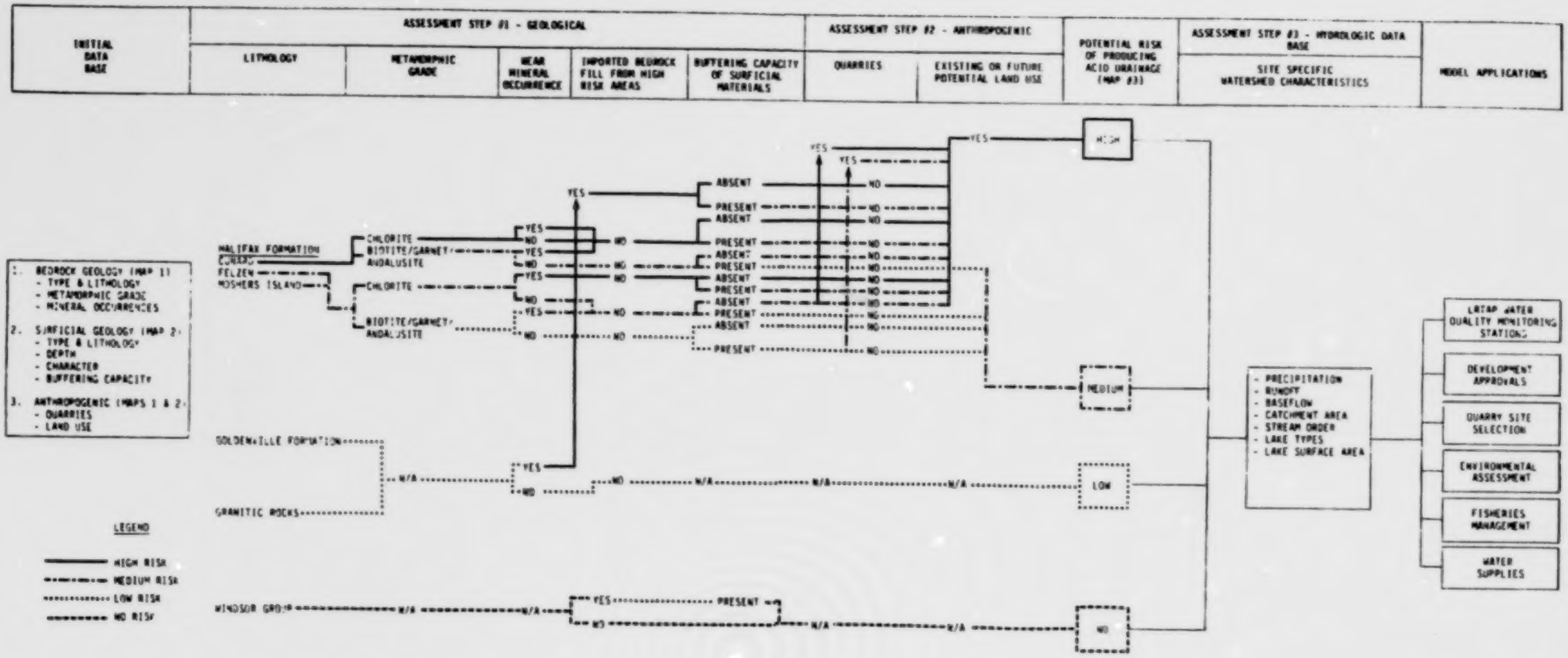
INITIAL DATA BASE
ASSESSMENT STEP #1 - GEOLOGICAL
LITHOLOGY
METAMORPHIC GRADE
NEAR MINERAL OCCURRENCE
IMPORTED BEDROCK FILL FROM HIGH RISK AREAS
BUFFERING CAPACITY OF SURFICIAL MATERIALS
ASSESSMENT STEP #2 - ANTHROPOGENIC
QUARRIES
EXISTING OR FUTURE POTENTIAL LAND USE
POTENTIAL RISK OF PRODUCING ACID DRAINAGE (MAP #3)
ASSESSMENT STEP #3 - HYDROLOGIC DATA BASE
SITE SPECIFIC WATERSHED CHARACTERISTICS
MODEL APPLICATIONS
1. BEDROCK GEOLOGY (MAP 1)
- TYPE & LITHOLOGY
- METAMORPHIC GRADE
- MINERAL OCCURRENCES
2. SURFICIAL GEOLOGY (MAP 2)
- TYPE & LITHOLOGY
- DEPTH
- CHARACTER
- BUFFERING CAPACITY
3. ANTHROPOGENIC (MAPS 1 & 2)
- QUARRIES
- LAND USE
HALIFAX FORMATION
CUNARD
FELZEN
MOSHERS ISLAND
CHLORITE
BIOTITE/GARNET/ANDALUSITE
GOLDENVILLE FORMATION
GRANITIC ROCKS
WINDSOR GROUP
YES
NO
ABSENT
PRESENT
N/A
HIGH
MEDIUM
LOW
NO
- PRECIPITATION
- RUNOFF
- BASEFLOW
- CATCHMENT AREA
- STREAM ORDER
- LAKE TYPES
- LAKE SURFACE AREA
LRTAP WATER QUALITY MONITORING STATIONS
DEVELOPMENT APPROVALS
QUARRY SITE SELECTION
ENVIRONMENTAL ASSESSMENT
FISHERIES MANAGEMENT
WATER SUPPLIES
LEGEND
HIGH RISK
MEDIUM RISK
LOW RISK
NO RISK

FIGURE 8 - MODEL FOR ASSESSING RISK OF ACID DRAINAGE (Porter Dillon, 1987)

problem of acid mine drainage in the context of water resources management is also highly variable, complex, and unpredictable. The generation of acid drainage and the liberation of metals from exposed sulphide bearing Meguma rocks in Nova Scotia is a problem for water managers both from a point source as well as a non-point source of pollution. These are the dimensions of the problem of acid mine drainage that are coming to light and have to be dealt with. To address the water management aspects of acid mine drainage from the Meguma, the following is required:

- a good understanding of not only where the Meguma slates occur, but the depth of overburden in those areas, and the nature and degree of mineralization in the units that contribute a high acid producing potential.
- evaluation of the sensitivity of the critical areas where the risk of acid mine drainage is high and there are particular concerns related to the water resources.
- co-operative and co-ordinated efforts among the various municipal, provincial, and federal government agencies that have expertise in dealing with the problem and collectively managing our water resources.
- general public information and education to raise the awareness of the problem and identification of ways to prevent generation of acid mine drainage.

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Electromagnetic Conductivity Surveys to Identify Acid Sources and Flow Patterns in U.S. Surface Coal Mines

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Abstract

An electromagnetic induction (EMI) method of measuring earth conductivity was applied at several backfilled surface coal mines to determine the utility of this method for acid mine drainage (AMD) assessments. The method was found to be a quick, useful tool for subsurface characterization, which is necessary for the design of effective at-source AMD abatement strategies. Survey results and interpretations along with the results of confirmatory drilling are presented.

The EMI method delineated major subsurface anomalies such as disturbed rock-undisturbed rock interfaces (highwall, sidewall, and low wall), variations in the mine floor, and changes in the saturated spoil thickness; such information facilitates mapping of generalized flow patterns. Localized zones of highly pyritic material were easily observed as high-conductive anomalies. The degree to which EMI data could be interpreted was site-specific and dependent on total thickness of backfill, relative thickness of saturation zone, and electrical properties of the backfill.

Introduction

The rock overlying coal seams in the Appalachian coal region may contain significant amounts of pyrite, particularly in Pennsylvania, Maryland, Ohio, Kentucky, and West Virginia. Surface coal mining in this area exposes pyrite to oxygen and water with the ensuing oxidation of pyrite resulting in acid mine drainage (AMD). AMD is characterized by low pH and contains high concentrations of acidity, sulfate, iron, and other metals, particularly manganese and aluminum.

U.S. Federal regulations require coal mine operators to treat acidic discharges to meet effluent standards for pH, acidity, iron, manganese, and suspended solids (U.S. Code of Federal Regulations, 1985). Treatment of AMD commonly consists of neutralization with hydrated lime or sodium hydroxide, aeration (natural or mechanical), and settling. Equipment, labor, and chemical costs, as well as costs associated with sludge handling and disposal, can make the treatment of AMD expensive. Treatment becomes an increasingly unattractive option as mines approach closure and revenue from coal sales ends.

As an alternative to treatment, at-source abatement techniques are currently receiving considerable attention. These at-source techniques attempt to mitigate AMD by reducing water and/or oxygen contact with the acid producing material or inhibiting iron-oxidizing bacteria which catalyze acid production. A critical first step in the design of an effective at-source abatement program at reclaimed surface mines is site-specific identification of acid sources and characterization of contaminant movement through the spoil. Random drilling of observation wells near and up-gradient from the seeps provides only localized information, which often does not fully characterize subsurface conditions.

Surface geophysics can permit preliminary characterization of the subsurface ground water environment. Electrical resistivity (ER) techniques are commonly used in ground water contamination studies (Zohdy, et al., 1974; Yazicigil and Sendlein, 1982) and have been applied to mine drainage problems (Merkel, 1972; Oteri, 1981).

Electromagnetic induction (EMI) techniques have long been used in mineral exploration, but only recently have advances in instrumentation made the technology readily adaptable to ground water pollution studies (Greenhouse and Slaine, 1982; Noel _et al_., 1982; McNeill, 1980; Duran, 1982). This report contains examples of previously reported and new U.S. Bureau of Mines field studies.

Geophysical Methods and Field Applications

The EMI method involves the use of a transmitter, transmitting coil, receiver, receiving coil, and connecting wire. An alternating current in the transmitting coil (wire loop) produces the primary magnetic field, H_p. This primary magnetic field induces electric fields, the strengths of which are dependent on earth conductivity. These electric fields in turn induce a secondary magnetic field, H_s. The receiving coil measures the ratio of $H_s:H_p$, which is linearly proportional to the apparent conductivity (σ_a). Apparent conductivity is defined by

$$\sigma_a = \frac{2}{\pi f \mu_o S^2} \frac{H_s}{H_p},$$

where f = frequency (Hz),

μ_o = permeability of free space,

and S = intercoil spacing (m).

The EMI instrumentation used in this study is entirely portable, requiring two people for field operations. A direct reading of conductivity in mmhos/m is obtained while the transmitting and receiving coils are held in a coplanar orientation (either horizontal or vertical plane) at one of three possible coil separations: 10 m, 20 m, or 40 m.

The transmitter operates at a fixed frequency for each of the three coil separations. Effective depth of penetration increases with increasing coil separation. Depth of penetration, as well as current distribution, can also be varied by alternating between the horizontal and vertical coil orientations (table 1). Restriction to only three coil separations and the associated limitation on quantitative vertical sounding capabilities are disadvantages of the EMI terrain conductivity technique.

Table 1. Effective depth of penetration for allowable coil separations and orientations (McNeill, 1980)

Coil separation (m)	Effective depth of penetration (m)	
	Vertical coils (horizontal dipoles)	Horizontal coils (vertical dipoles)
10	7.5	15
20	15	30
40	30	60

More detailed descriptions of terrain conductivity and principles of operation in ground water studies are given in McNeill (1980) and Stewart (1982). Theoretical discussions of electromagnetic induction can be found in standard geophysical texts (Keller and Frischknecht, 1966; Grant and West, 1965).

One application of geophysical surveying at surface mines is to determine if any localized sources of acid generation exist within the spoil material. Localized sources may result from the selective burial of pyritic material within the spoil, such as coal cleanings, segregated overburden strata, or preparation plant refuse. Often these materials contain a higher percentage of pyritic sulfur than the bulk spoil material and, unless completely isolated from sources of water and/or oxygen, can contribute greatly to AMD loading. The existence and location of these pods of buried material are rarely documented, particularly at old or abandoned mine sites. Field studies at sites SM1 and SM2 illustrate the use of EMI equipment to locate buried pockets of acid-producing material.

A second application is to define general flow patterns through the spoil. Spoil water flow is largely dictated by boundary conditions, the most prominent of which are flow barriers at interfaces between spoil and undisturbed rock strata. Lateral boundaries include any highwall, sidewall, or outcrop barrier (low wall) that may exist at the site. The lower flow boundary is usually the mine floor. All boundaries may be completely buried during regrading and reclamation. Field studies at sites SM3, SM4, and SM5 illustrate the use of the EMI equipment to define flow boundaries. Spoil water flow may also be influenced by mine floor contours. EMI equipment can locate this channelized flow, as evidenced by field study at site SM6.

Field Investigations

SM1

Site SM1 is a 6-hectare (ha) subarea of a 32 ha surface mine in Upshur County, WV. The site was mined for the Lower and Middle Kittanning coal seams in the mid-1970's and reclaimed shortly after mining was completed. Reclamation included grading and revegetating, but the site was not returned to approximate original contour. As a result, present site topography consists of a steep slope at the old highwall, a broad, flat bench over the mined area, and a steep outslope (Figure 1). The mine floor dips slightly to the northwest, and maximum spoil thickness on the bench area is about 8 m. Following reclamation, a perennial acidic seep with mean annual discharge of 1.1 l/s (18 gal/min) developed at the toe of the outslope.

EM surveying of the site was done to facilitate field application of a bactericidal agent to decrease acid production (Kleinmann and Erickson, 1983). This method of abatement requires direct application of the bactericide to the acid-producing zones, so it was of considerable importance to determine the location of any pockets of pyritic material or localized zones of poor water quality.

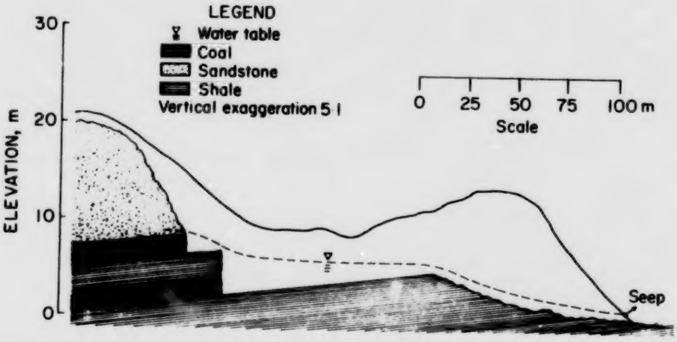

Figure 1. Cross-section of site SM1.

Contoured results of a rectangular grid of EMI profiles obtained using a 10-m intercoil spacing and horizontal dipole are shown in Figure 2. The plot is a composite of two surveys run perpendicular to each other. The value at each grid point represents the average of the two readings obtained at that point. The averaging was used to smooth some of the anisotropic effect caused by alignment of the transmitting and receiving coils with a linear conductive body. Each survey resulted in collection of 60 data points and required approximately 5 hours of field time for a 2-person crew. The locations of eight observation wells installed subsequent to the surveying are also shown in Figure 2.

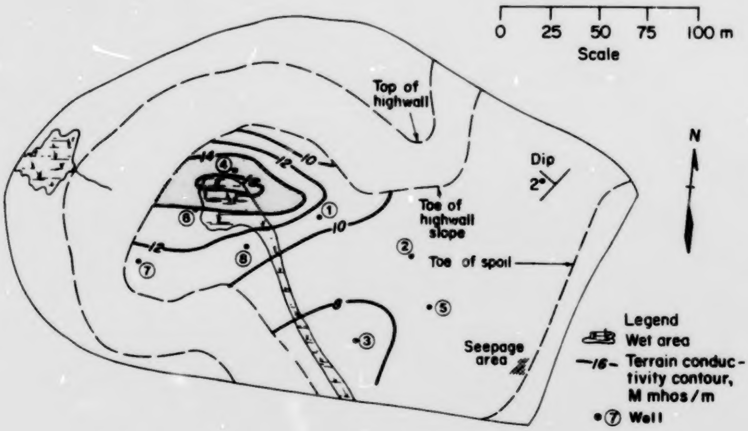
0
25
50
75
100 m
Scale
Top of highwall
N
Dip
2°
14
12
10
16
④
①
⑥
10
Toe of highwall slope
12
⑦
⑧
②
Toe of spoil
8
⑤
③
Seepage area
Legend
Wet area
16 Terrain conductivity contour, M mhos/m
⑦ Well

Figure 2. Contoured plot of apparent conductivity at site SM1. Readings were obtained using the 10-m intercoil spacing and horizontal dipole. Contour interval is 2 mmho/m. Shaded area is area of highest apparent conductivity.

Apparent conductivities at the site ranged between 6 and 18 mmho/m, the highest values occurring in the northwestern (upper left) corner. The highest apparent conductivities (14 to 18 mmho/m) appear to be related to a fine-grained, carbonaceous material (coal cleanings) buried in the spoil. Drill holes 4 and 6, both located in areas with apparent conductivities greater than 14 mmho/m, contained 6 m and 2 m of the cleanings, respectively. Spoil in the six other drill holes contained only a brown, weathered sandstone with particle sizes ranging from clay to small boulders. Laboratory analysis performed on samples obtained from the drill holes determined that the cleanings contained a higher percentage of sulfur than the weathered sandstone, with mean values of 1.24 and 0.12 pct, respectively. Samples of the underclay were also analyzed and exhibited a mean sulfur content similar to that of the cleanings, 1.20 pct.

The lowest apparent conductivities on the site were observed in the southeastern (lower right) corner and are interpreted as the spoil outslope. The outslope is generally composed of very coarse boulders, has a steep hydraulic gradient, and contains little stored spoil water. The coarse material rapidly transmits overflow from the mine floor to the discharge area. Drill holes 2, 3, and 5, all located in areas with apparent conductivities less than 10 mmho/m, are all believed to be on the outslope. Wells 3 and 5 are dry at all times of the year, while well 2 is intermittently dry.

Not surprisingly, the poorest water quality on the site was observed at well 4. Spoil water at well 6 was slightly less contaminated as a result of recharge from relatively clean highwall seepage on the southwestern part of the site near well 7. Areal variations in water quality were impossible to determine from the raw EMI data, owing to the electrical heterogeneity of the spoil material. Similarly, temporal variations in water quality and quantity were not readily evident from a replicate EM survey conducted in spring at high water levels.

SM2

Site SM2 is a 6-ha section of a larger site located in Clarion County, PA. The site was mined for the Upper and Lower Clarion coal seams in the early 1970's. Both the Lower Clarion highwall and the outcrop barrier remain intact (Figure 3). Spoil thickness ranges from 2 m near the outcrop barrier to more than 20 m near the Lower Clarion highwall. The Upper Clarion coal was removed from the entire 6-ha area, resulting in slightly more than 7 m of spoil over the highwall bench. The mine floor dips gently to the southwest. The spoil is predominantly sandstone overburden but contains pockets of buried preparation plant refuse. Leaching tests indicate the refuse is highly acid producing.

The site is drained by a small stream flowing to the southwest. The ground surface slopes toward the stream at an average 7 pct grade. A perennial seep issues from the soil near the stream, discharging approximately 0.13 l/s (2.0 gal/min). The seep discharge is characteristic of AMD, containing 5,000 mg/l sulfate, 4,000 mg/l acidity, 800 mg/l iron, and 275 mg/l aluminum.

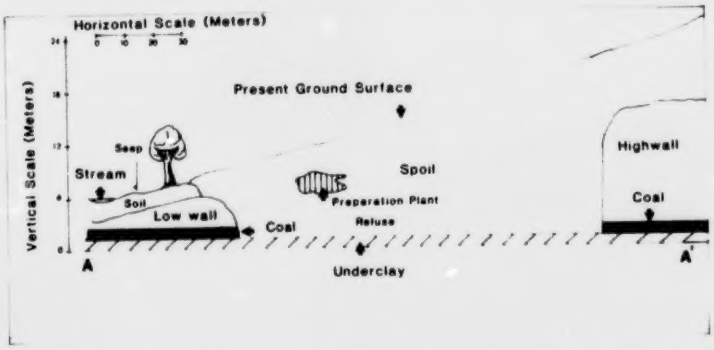
Horizontal Scale (Meters)
0
10
20
30
Vertical Scale (Meters)
24
18
12
6
0
Present Ground Surface
Seep
Stream
Soil
Low wall
Coal
Spoil
Preparation Plant
Refuse
Highwall
Coal
A
A'
Underclay

Figure 3. Cross section of site SM2. A-A' location is shown on figure 4.

Ten EMI profiles were conducted along the site contour at N 80° E, using the vertical coil orientation and both 10-m and 20-m intercoil spacing (Figures 4 and 5). Readings were taken on 10 m centers, resulting in 200 data points for each intercoil spacing. The complete survey took 2 days. Drilling logs from eight existing observation wells (A-H) and five new observation wells (T, U, X-Z) were used for correlation with the EMI results. DC resistivity soundings were also used to estimate vertical layering and true resistivities of the spoil and bedrock.

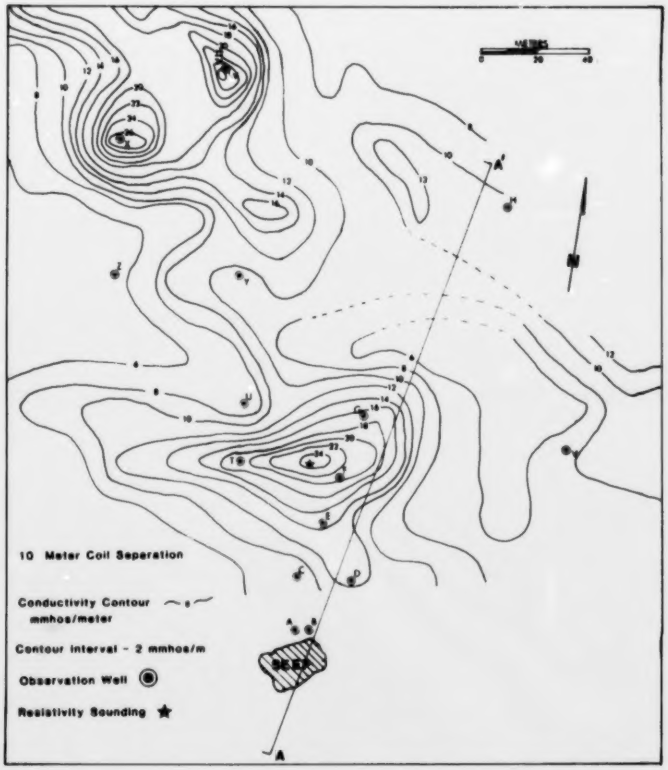
METERS
0
20
40
N
A'
A
SEEP
10 Meter Coil Seperation
Conductivity Contour — 8 —
mmhos/meter
Contour interval - 2 mmhos/m
Observation Well ◉
Resistivity Sounding ★

Figure 4. Contoured plot of apparent conductivity at site SM2. Readings were obtained using the 10-m intercoil spacing and horizontal dipole.

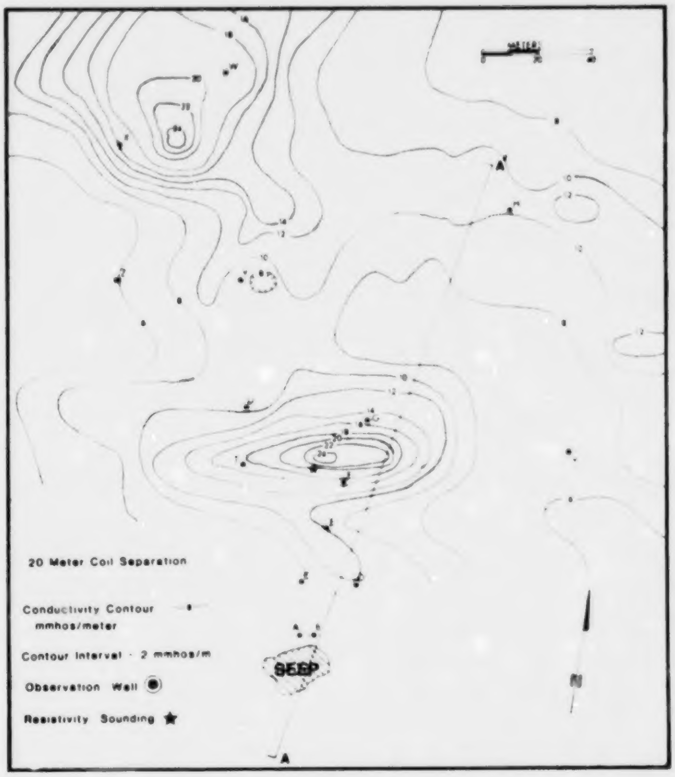
20 Meter Coil Separation
Conductivity Contour mmhos/meter
Contour Interval · 2 mmhos/m
Observation Well
Resistivity Sounding
SEEP
A
N

Figure 5. Contoured plot of apparent conductivity at site SM2. Readings were obtained using the 20-m intercoil spacing and the horizontal dipole.

A contour map of apparent conductivity for the 10-m coil separation is shown in Figure 4. The most conspicuous features are the two high-conductivity areas, with values exceeding 20 mmho/m. The high zones are well defined and are outlined by relatively steep electrical gradients. The remainder of the site had apparent conductivities less than 14 mmho/m.

Both high-conductivity zones were associated with pockets of preparation plant refuse buried in the predominantly sandstone spoil. Wells F, G, and X (Figure 4) were the only borings to encounter the refuse, and all three are at or near the centers of the high zones (well W was not drilled). At wells F and X, the refuse is about 3 m below the ground surface. At well G, a very thin layer of refuse was encountered 10 m below the surface. Apparent conductivities obtained using the horizontal dipole reflect a proportionally greater contribution from near-surface material (McNeill, 1980), resulting in the higher conductivities for the shallow refuse near F and X. At the 10-m coil separation, more than 75 pct of the observed apparent conductivity can be attributed to the material within 10 m of the surface (McNeill, 1980).

As at the SM1 site, saturated water quality effects are small relative to the variable conductivity of the spoil and/or refuse. The water table at well F is about 3 m below the surface, but the spoil at well X is unsaturated to at least 16 m below the surface. Because the saturated zone is well below the optimum depth range of the 10-m intercoil spacing, variations in saturated zone water quality can be eliminated as a cause of the conductivity anomaly at well X. This contention is further supported by a higher degree of resolution obtained at the 10-m intercoil spacing as compared to the 20-m spacing (Figure 5), indicating the conductor occurs at shallow depth.

SM3

Site SM3 is a 52-ha clay surface mine in Fayette County, PA. The clay underlying the Lower Kittanning coal seam was mined from the mid-1950's through the early 1970's. All of the mining occurred along a narrow ridge with no known outcrop barriers left intact. The site was partially regraded and revegetated near the old highwall in the mid-1970's and subsequently developed a perennial acidic seep with a mean annual discharge of 1.9 l/s (30 gal/min) into a nearby trout stream. Thickness of the spoil ranges from 20 m to 34 m, and the mine floor dips 5° to the northwest.

Part of the abatement plan at SM3 called for a series of wells to inject alkaline fluid directly into the spoil for in-place neutralization. Determination of the recharge area and general flow pattern was necessary for siting the injection wells. As highwall seepage was believed to be a major source of recharge to the spoil, accurate location of the buried highwall was required. A series of EMI profiles using the 20-m intercoil spacing and horizontal dipole were conducted on an 8-ha area above the discharge. The survey resulted in 140 data points and required 8 hours of field time for a 2-person crew to complete.

Contoured results of the survey are shown in Figure 6. A wide range of conductivities were observed, with values ranging from 4 to 34 mmho/m. Of greatest interest in highwall location are the set of linear contours ranging from 6 to 14 mmho/m on the right side of the plot. The linearity of the contours and the steep electrical gradient imply a uniform electrical boundary interpreted as the highwall. A series of closely spaced EMI readings at various bearings were used to refine the interpretation in the field.

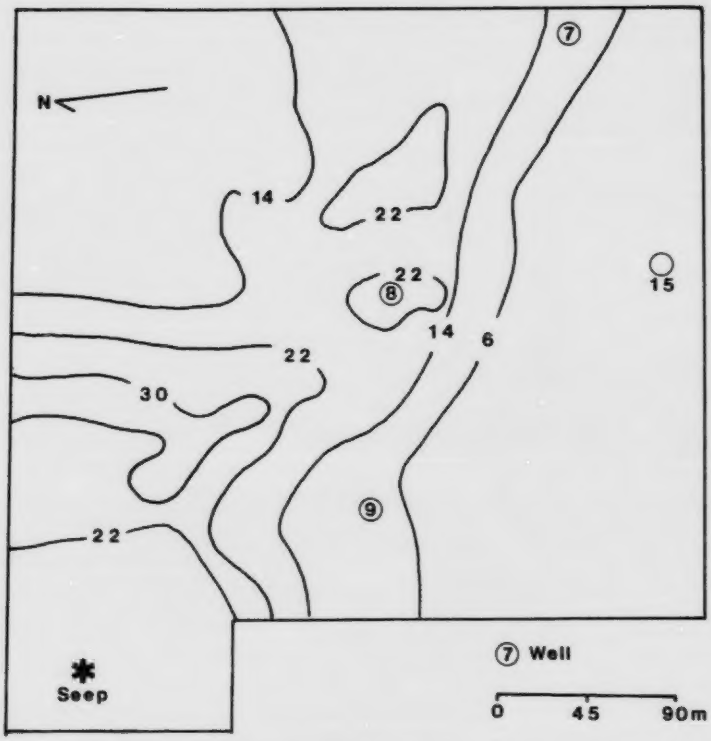

Figure 6. Contoured plot of apparent conductivity at site SM3. Readings were obtained using the 20-m intercoil spacing and horizontal dipole. Contour interval is 8 mmho/m.

Based on the EMI survey, the series of observation wells shown in Figure 6 were proposed. Wells 7, 8, and 9, drilled just inside the interpreted spoil-highwall interface, all encountered over 20 m of spoil material above the mine floor. Well 15 was drilled on top of the buried highwall and encountered undisturbed rock below a thin spoil cover. A series of 10 shallow borings drilled on a line between wells 7 and 15 confirmed the actual position of the old highwall at approximately 10 m southwest of well 7. Using the EMI data, the extension of the highwall can be located relatively accurately. Flow of spoil water downdip along the highwall is inferred from the location and strike of the highwall relative to the discharge area.

SM4

SM4 is an 8-ha abandoned surface mine in Butler County, PA. The site is believed to have been mined in the 1950's, but detailed records are not available. In 1981, the mine was reclaimed under the Rural Abandoned Mine Program administered by the Soil Conservation Service. The spoil has a maximum thickness of 7 m near the old highwall on the western boundary of the site and grades rapidly to less than 2 m in the central and eastern parts of the site. A limestone-lined subsurface drain was installed along the foot of the highwall to intercept and transport highwall seepage north to a rock-lined waterway (Figure 7). Since completion of the project, discharge from the subsurface drain has been slightly acidic (acidity 20 mg/l, iron 18 mg/l, sulfate 200 mg/l). In addition, an intermittent acidic seep has developed on the southern end of the site.

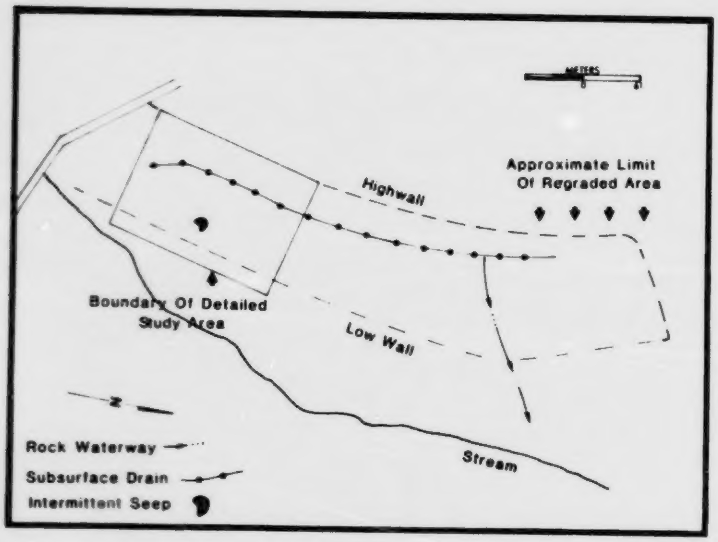
METERS
Approximate Limit Of Regraded Area
Highwall
Boundary Of Detailed Study Area
Low Wall
Stream
N
Rock Waterway
Subsurface Drain
Intermittent Seep

Figure 7. Diagram of site SM4 showing location of detailed study area.

Initially, three EMI profiles oriented parallel to the highwall were conducted across the length of the site using 10-m intercoil spacings. The resulting contour plot was nondescript except for an apparent conductivity high centered on the southern end of the site near the intermittent seep. Ten EMI profile lines were run perpendicular to the highwall on a 1.6 ha area near the intermittent seep using the 10-m intercoil spacing. The survey resulted in 60 data points and required 6 hours of field time for a 2-person crew.

The contoured results of the 10 profiles are shown in Figure 8. Apparent conductivity ranged from 14 to 34 mmho/m. The lowest apparent conductivities were observed along the eastern and western boundaries of the site, ranging from 14 to 24 mmho/m. The boundaries are marked by closely spaced, linear contours and are interpreted as the highwall and low wall. The central region of the study area is characterized by several pockets of varying conductivity, ranging from lows of 24 mmho/m to highs centered at 30 to 34 mmho/m.

The conductivity high (34 mmho/m) centered in the northwestern corner of the study area appears to project to the south along the base of the highwall contours. Although not sharply defined, the projection is interpreted as associated with the subsurface drain. Similarly, a zone of high apparent conductivity (greater than 30 mmho/m) extends along the base of the low wall contours, separated into two centers by a ridge of lower apparent conductivity (less than 28 mmho/m). The high conductivity along the low wall is interpreted as ground water within the spoil. The low-conductivity ridge is interpreted as a high point in the bedrock, due to either a roll in the mine floor or an unmined block of coal. The existence of an unmined block of coal of undetermined dimensions was confirmed by conversations with inspectors who were on site during regrading of the spoil.

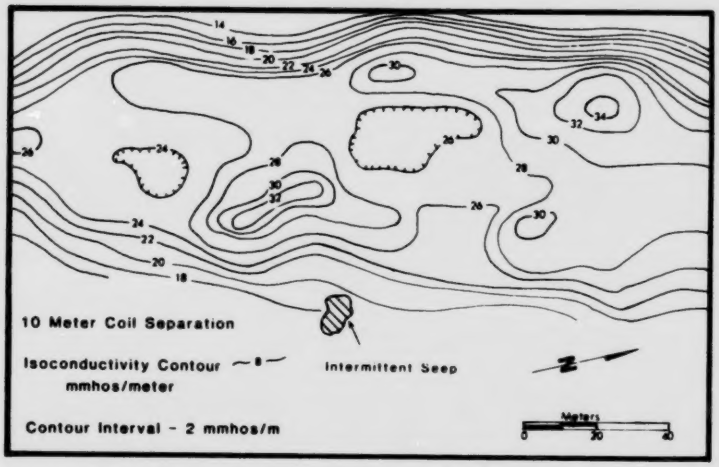
14
16
18
20
22
24
26
30
34
32
30
26
24
28
30
32
26
28
26
30
24
22
20
18
10 Meter Coil Separation
Isoconductivity Contour ~8~
mmhos/meter
Intermittent Seep
N
Contour Interval - 2 mmhos/m
Meters
0
20
40

Figure 8. Contoured plot of apparent conductivity at the detailed study area of site SM4 as shown in figure 6. Readings were obtained using the horizontal dipole.

A schematic diagram of the physical model developed from this interpretation is shown in Figure 9. Seepage from the highwall is intercepted by the subsurface drain and transported north to the drain outlet. The bedrock high acts a barrier to ground water flow in the southern section of spoil, resulting in subsurface storage south of the barrier. During recharge events, water levels rise and the stored water discharges to the surface at the intermittent seep. North of the barrier, ground water flows along the low wall and discharges near the rock-lined waterway.

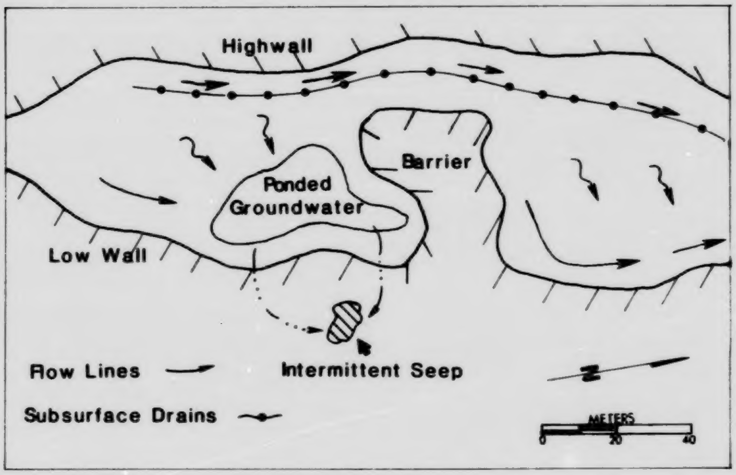

Figure 9. Physical interpretation of the apparent conductivity distribution shown in figure 7.

SM5

Site SM5 is an 8-ha subarea of a 30-ha reclaimed surface mine located in Clearfield County, PA (Figure 10). Part of the eastern half of the site was mined for the Lower and Middle Kittanning coal seams in the 1950's. This part of the site remained abandoned until 1976. From 1976 to 1980, the remaining coal on the eastern half of the site was mined with subsequent reclamation to approximate original contour. The western half of the site was mined between 1980 and 1983. Unlike the eastern half (area to the right of the mining company boundary in figure 10), where all of the coal was removed, undisturbed barriers of intact strata (low wall and sidewall) were left along the southern and western limits of the site. This half of the site was mined by a different company and was returned to approximate original contour and revegetated in 1983. The mine floor dips slightly to the south, and spoil thickness ranges from 10 m at the southern end of the site to 18 m at the northern end. Following reclamation, two perennial acidic seeps with total mean annual discharge of 0.4 l/s (6.5 gal/min) developed. These seeps discharge AMD containing 2,500 mg/l sulfate, 800 mg/l acidity, 110 mg/l iron, and 150 mg/l manganese.

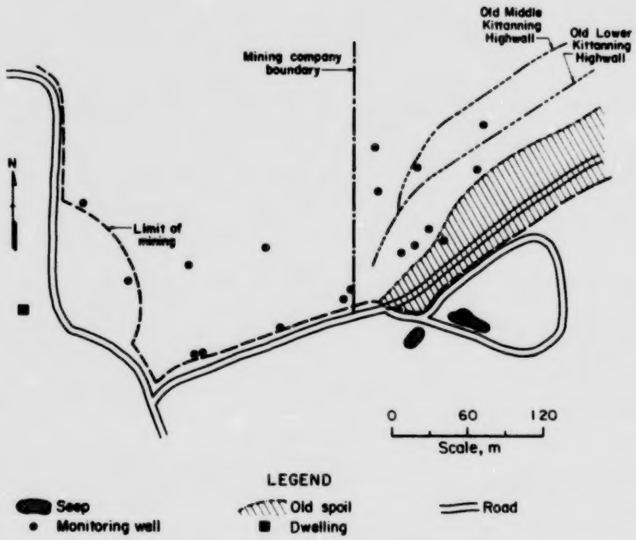
Old Middle Kittanning Highwall
Old Lower Kittanning Highwall
Mining company boundary
N
Limit of mining
0
60
120
Scale, m
LEGEND
Seep
Monitoring well
Old spoil
Dwelling
Road

Figure 10. Diagram of site SM5.

The site was surveyed with the EMI equipment to investigate the applicability of various at-source abatement techniques. The mining companies were particularly interested in alkaline injection for AMD mitigation. The use of this abatement technique requires information about recharge areas and generalized flow patterns. Also, the location of any pockets of highly pyritic material may suggest possible alternative abatement techniques.

Contoured results of 13 EMI profiles obtained using a 20-m intercoil spacing and horizontal dipole are shown in Figure 11. The entire 8-ha survey resulted in 194 data points and took a 3-person crew 8 hours to complete.

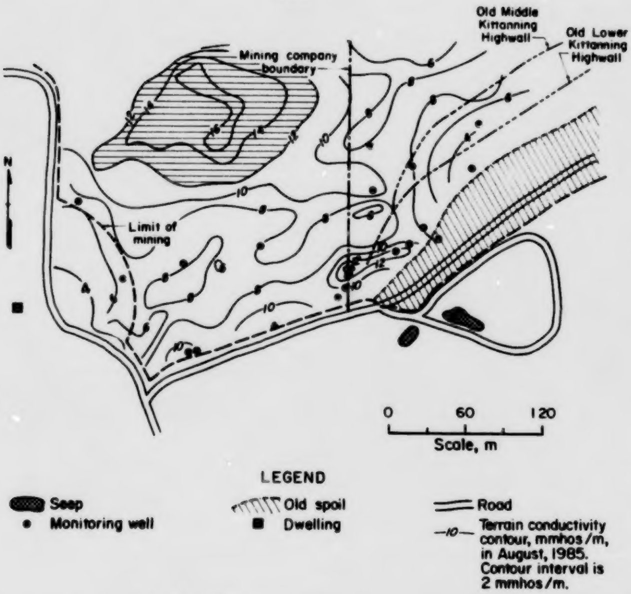
Old Middle Kittanning Highwall
Old Lower Kittanning Highwall
Mining company boundary
Limit of mining
N
16
14
12
10
8
6
4
0 60 120
Scale, m
LEGEND
Seep
Monitoring well
Old spoil
Dwelling
Road
—10— Terrain conductivity contour, mmhos/m, in August, 1985. Contour interval is 2 mmhos/m.

Figure 11. Contoured plot of apparent conductivity at site SM5. Readings were obtained using the horizontal dipole. Horizontally shaded area is area of highest apparent conductivity.

Apparent conductivities at the site ranged between 4 and 16 mmho/m, the lowest values occurring at the eastern and western ends of the site. The low apparent conductivities (<6 mmho/m) at the eastern end of the site are associated with dry conditions, as evidenced by low water levels in the monitoring wells. The area of low conductivity (<6 mmho/m) at the western end of the site is related to an unmined area. U.S. Federal law prohibits mining within a 46-m radius of dwellings. The 6 mmho/m contour line corresponds quite well to this limit of mining.

The only wells that contained an appreciable amount of water were located at the southern end of the site, directly upgradient from the low wall. Water in wells in this area was only 3 to 4 m below the ground surface and extended to the mine floor, creating a saturation zone 4 to 5 m thick. Most of this area produced only slightly higher conductivities (>10 mmho/m); however, higher conductivities (10 to 14 mmho/m) were found above the eastern end of the low wall, where the quality of the water was the poorest.

The highest conductivity values (14 to 16 mmho/m) were observed in the northwestern area of the site. This pattern of higher conductivity values is typical of the pattern produced by a pocket of a material with high pyritic content. No confirmatory drilling was done in this area.

SM6

Site SM6 is located in Clinton County, PA. The mine operated in the mid-1970's and was reclaimed in 1979. The reclaimed area consists of three lobes (northern, southern, and eastern) with an areal extent of 15 ha and a maximum spoil thickness of 10 m. The mine pavement dips gently to the southeast.

Two areas of perennial seepage occur near the southern and eastern (downdip) lobes. At the time of the study, seep discharge contained 1,100 mg/l acidity, 600 mg/l total iron, and 1,800 mg/l sulfates. The discharge is intercepted by two limestone-lined drainage ditches and transported to a series of oxidation ponds. Although discharge rates were not measured, workers familiar with the site noted the rate was relatively low at the time of the study (December).

EMI surveys of the site at both 10- and 20-m intercoil spacings were completed in 3 days. (Frozen ground and a light snow cover would have made DC resistivity surveying at the site difficult.) Figure 12 is a contoured plot of the apparent conductivity data from 10 survey lines oriented at N 65° W using 20-m intercoil spacings.

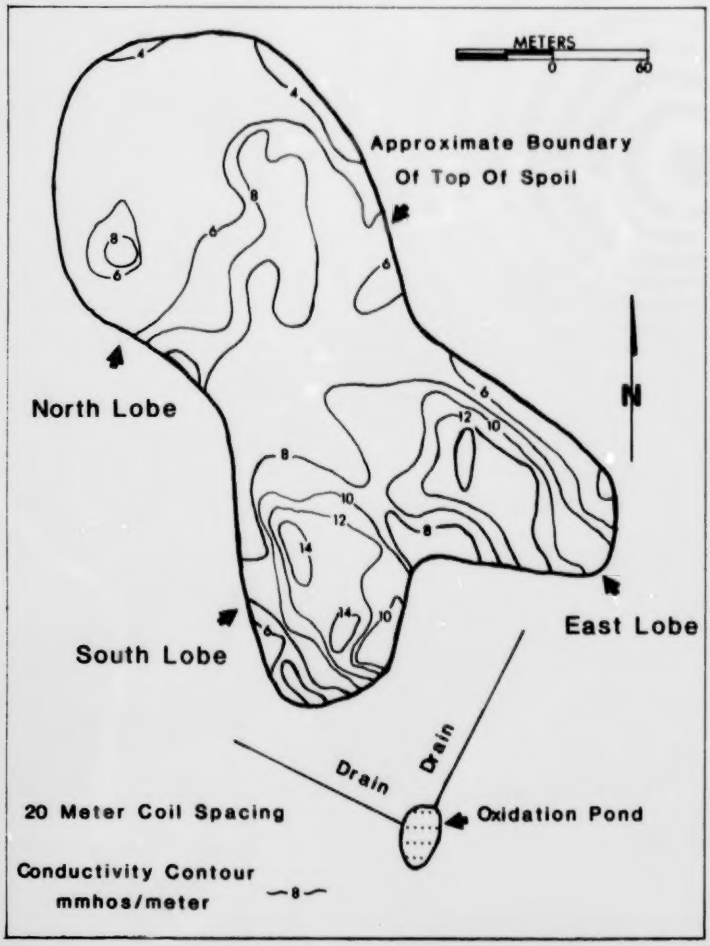
METERS
0
60
4
4
Approximate Boundary
Of Top Of Spoil
8
6
8
6
6
North Lobe
N
6
12
10
8
10
12
8
14
14
10
6
South Lobe
East Lobe
Drain
Drain
20 Meter Coil Spacing
Oxidation Pond
Conductivity Contour
mmhos/meter
8

Figure 12. Contoured plot of apparent conductivity at site SM6. Readings were obtained using the horizontal dipole.

The northern lobe of spoil is characterized by relatively low apparent conductivity (<10 mmho/m), with no distinct electrical features. Apparent conductivities were higher on the southern and eastern lobes, ranging between 8 and 16 mmho/m. The conductivity highs were elongated in a southeasterly direction to the edge of each lobe.

An additional series of five EMI survey lines were conducted on the southern lobe using the 10-m intercoil spacing. The survey lines resulted in a similar, but slightly more pronounced, distribution of apparent conductivity values, ranging as high as 20 mmho/m. The high was again elongated in a southeasterly direction to the edge of the spoil.

Interpretation of the results was aided by the drilling of wells subsequent to the EMI work. The apparent conductivity highs were associated with an increase in the thickness of saturated materials. Wells situated near the centers of the apparent conductivity highs intersected the water table above the pavement floor. Wells installed away from these high zones did not.

The greatest volume of stored water is located in the downdip section of the spoil, near the seepage discharge points. The storage of ground water in sections of each lobe is associated with depressions in the mine pavement. The high-apparent-conductivity zones are elongated downdip, in the direction of flow to the seeps. The intersections of the highs with the edge of the spoil are interpreted as primary seepage discharge points.

Discussion

Initially, the approach to using geophysical surveys at surface mines was similar to that of plume identification at landfill sites. The surveys were designed to determine spoil water flow patterns and major acid-production zones by defining a "plume" of acid-contaminated water and outlining its origin or area of maximum concentration. EMI surveying at a dozen surface mines, including the six described in this paper, has resulted in a slight alteration of this approach. Electrical identification of a "plume" of water quality is difficult due to heterogeneity in the electrical properties and flow characteristics of the spoil. Although the spoil is treated as a single electrical unit relative to the undisturbed rock, considerable variations in properties occur as a result of the method of spoil placement and both intentional and unintentional material segregation. As a result, a more reasonable approach to geophysical characterization of the spoil water quality and flow regime was found to be direct identification of potential sources of acid production and definition of flow boundaries.

Sources of acid production were easily identified at mine sites SM1 and SM2. Pockets of preparation plant refuse and coal cleanings with a higher acid-producing potential than the bulk spoil were associated with the highest apparent conductivities at both sites. The high conductivity of the cleanings and refuse may be a function of the fine-grained nature of the material, unsaturated moisture retention properties, unsaturated moisture quality, or mineral assemblages.

Location of these pockets of potentially acid-producing material is of primary importance to the development of an at-source abatement plan.

Lateral flow boundaries are also readily evident from EM survey data. The boundaries, including highwalls, sidewalls, mining relicts, and outcrop barriers (low walls), are marked by sharp electrical gradients along a linear trend. As the bulk permeability of the spoil is usually higher than that of the undisturbed rock, the boundaries are barriers to spoil water flow and often determine storage zones and discharge locations. The positions of the lateral flow barriers combined with the approximate dip of the mine floor dictate the basic characteristics of the spoil flow regime.

Structural rolls in the mine floor (lower floor boundary) can further modify storage and flow characteristics. Defining structural rolls with electromagnetic data depends on the electrical properties of the mine floor and the spoil, total spoil thickness, and relative thickness of saturated and unsaturated zones in the spoil. In general, uniform spoils (no pockets of anomalously-high-conductivity material) exhibit higher apparent conductivities over structural depressions than over structural rises. The higher apparent conductivity is ascribed to increased saturated storage in the depression. Structural rises on two of the sites surveyed act as spoil water divides, separating flow on different parts of the sites. Successful mapping of storage zones and mine floor structure is generally limited to sites with total spoil thickness of less than 20 m and saturated-unsaturated spoil ratio of at least 1:3.

For work at surface mines, commercially available EMI equipment provided several advantages over the more commonly used resistivity equipment, particularly for preliminary surveying at large sites. The most significant advantage was speed of field operations. At two sites where both EMI and ER were used, the ER survey took approximately four times longer than a comparable EMI survey. A second advantage was portability, particularly in dense vegetation or rocky terrain. The EMI equipment does not require ground contact or handling of multiple spools of wire. This was especially important when surveying along overgrown stream valleys, at surface mine spoils lacking topsoil, and on snow-covered or frozen ground.

The primary disadvantage of the EMI equipment was the lack of flexibility. The limitation of the EMI equipment to only three coil spacings and two coil orientations renders it much less amenable to vertical soundings and quantitative interpretation than ER. For general profiling across a large site, this was not a significant disadvantage. At sites where more detailed information is required, broad EMI profiling combined with ER soundings in areas of interest may be the most efficient means of obtaining necessary data.

Acknowledgments

Field testing of the geophysical equipment would not have been possible without cooperation of the coal companies involved. The authors also with to thank Michael Gardner for his assistance in data collection and interpretation.

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WILL IT GENERATE AMD?

AN OVERVIEW OF METHODS TO PREDICT ACID MINE DRAINAGE

By K. D. Ferguson[1] and P. M. Erickson[2]

INTRODUCTION

Many believe that mining effluents always contain metals. While technically correct most mine effluents contain either high or low levels though the concentration and loading of metals varies from mine to mine. Acid mine drainage (AMD) is usually the cause of high metal concentrations in contaminated effluents. Mines that have AMD have difficulty in meeting environmental standards and usually expend large financial resources on pollution control. It is imperative that mining companies and regulatory agencies attempt to predict AMD prior to mining so that mines can be designed that are cost effective and compatible with the environment. If AMD is generated, then companies should identify the source so that mitigation measures can be accomplished for minimal cost. In this paper, we discuss the prediction of AMD prior to mining and the identification of acid sources after drainage has become contaminated.

Formation of AMD

Prediction of acid mine drainage must be based on a thorough knowledge of the process of formation and the factors that control it. Unfortunately, the process is complex and not completely understood despite several decades of intense study. The knowledge base is considerable, however, and growing as illustrated below.

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Kleinmann et al., 1981, identified three stages in the generation of acid mine drainage. In the first stage, chemical and, or, biological oxidation of pyrite and other sulfide minerals slowly produces acid (Figure 1). This acid may initially be neutralized by carbonate minerals in the rock. After the carbonates are consumed, the process enters the second stage. As the pH drops in the microenvironment around the sulfide minerals, populations of acid loving (acidophilic) bacteria such as Thiobacillus ferrooxidans begin to multiply. These bacteria cause a further decrease in pH, causing the process to enter the third stage. When the pH around the sulphides decreases to below 3, ferric iron remains in solution. Ferrous iron is converted by the bacteria to ferric iron which in turn oxidizes the sulphide minerals. The rate of acid formation is rapid in the last stage and is limited by the concentration of ferric iron.

The four equations in Figure 1 generally describe the acid production process with pyrite (FeS_2) as occurs at coal mines, but, describe only a few of the many sulphide oxidation reactions at metal mines. The mix of sulphide minerals at metal mines accounts for the diversity in AMD quality.

Ferguson and Erickson (1986) classified the factors that control AMD formation as primary, secondary, and tertiary. The primary factors are those directly involved in the acid production process. Secondary factors control the consumption or alteration of the products from the acid generation reactions, while tertiary factors are the physical aspects of the waste materials or minesite that influence acid production, migration, and consumption.

The primary factors include pyrite and other sulfide minerals, oxygen, water, ferric iron, and iron-oxidizing bacteria (Figure 1). The physical relationship of these factors in an acid generating mine spoil is shown in Figure 2. Here, water and oxygen are provided by the external environment to the acid generating sites within the spoil. The massive transfer of reactants supports rapid acid production.

Regardless of the presence of pyrite and other sulfide minerals, AMD may not be a problem at a minesite due to secondary factors. The most important secondary factor is the neutralization of acid by alkalinity released from carbonate minerals in the mine waste, such as calcite ($CaCO_3$) and dolomite [$CaMg(CO_3)_2$] as described previously. The neutralization by calcite of acidity produced from pyrite oxidation may be represented as follows (Williams et al., 1982):

$$FeS_2(s) + 2CaCO_3(s) + 15/4O_2(g) + 3/2H_2O \rightarrow Fe(OH)_3(s) + 2SO_4^{2-} + 2Ca^{2+} + 2CO_2(g)$$

Other secondary factors include alteration of oxidation products by further reaction, such as ion exchange on clay surfaces, gypsum precipitation, and acid-induced dissolution of other minerals.

The physical characteristics of a mining waste, the spatial relationship between wastes, and the hydrologic regime are some of the tertiary factors that affect the AMD process. The important physical characteristics include particle size, physical weathering tendency, and mine waste permeability. The rate of acid production is a function of the particle surface area since that property reflects the amount of sulphide exposed for reaction.

Water serves several roles in the acid production system, acting as a reactant, as a reaction medium, and as a product transport solvent.

The overriding principle in the generation of acid mine water is that if the alkalinity released by minerals or already present in the contacting water exceeds the acidity produced then the system remains neutral or basic and the process does not enter Stage II. In this case, the bacteria responsible for catalyzing the acid generation reactions are inhibited and the solubilities of ferrous and ferric iron are reduced - both factors inhibit acid production.

The prediction of acid mine drainage involves the consideration of the few important factors that might control the acid production and consumption reactions at a minesite and the synthesis into a qualitative statement forecasting conditions during and after mining. Models and testing techniques have been developed to aid investigations in making this forecast.

Pre-Mine Prediction of AMD

Methods to predict acid mine drainage can be divided into five groups (Ferguson and Erickson, 1986).

1. geographical mining comparisons
2. paleoenvironmental and geological models
3. geochemical static tests
4. geochemical kinetic tests
5. mathematical models

Geographical mining comparisons involve examining and sampling drainage from abandoned or operating mines near the project under study. Simple extrapolations are made for similar mining operations in related geological environments.

Paleoenvironmental models examine the regional depositional environment of the sulphide minerals. Pyritic sulphur formed in marine or brackish water may have a greater tendency to generate AMD than that formed in fresh water environments (Caruccio et al., 1977, Williams et al., 1982). Geological models of a sedimentary sequence or igneous/ metamorphic orebody are valuable in estimating the location and size of potentially acid generating zones.

Geochemical static tests simply compare the amount of potentially acid generating sulphides to acid consuming carbonates in a rock sample (Figure 3). The acid/base accounting and APP: Sulphur Ratio tests are used frequently for coal mines in the United States while the B.C. Research Initial Test is used frequently for metal mines in Canada. The acid potential determinations are similar for the three tests, but, the neutralization potentials are different. The acid/base account neutralization potential involves a stronger digestion of carbonate minerals and may produce a slightly higher value than the other two procedures (Ferguson, 1982, Caruccio et al., 1977).

The acid/base account includes a format for presenting AMD prediction data useful for surface coal mines disturbing many lithologic groups (Figure 4). The net neutralization potentials of the individual groups may be used to develop a waste management plan for attempting to prevent AMD formation.

These tests are called static as they do not consider the relative rates of acid production and consumption. The rate of release of acidity from sulphide oxidation, and alkalinity from carbonate dissolution may be quite different and could affect the generation of acid mine drainage. Geidel (1979) suggests that the release of alkalinity is limited to a maximum value by the partial pressure of carbon dioxide while the buildup of acidity is not constrained. Various investigators have criticized the use of static tests and suggested they are only applicable to obvious cases where the quantity of acid producing or acid consuming material is in significant excess (Caruccio, et al., 1980; Williams et al., 1982; SENES, 1984). However, Ferguson and Erickson (1986) in their review of AMD prediction techniques, found that static tests were often accurate in predicting drainage quality and were particularly valuable as screening tests to determine if more sophisticated procedures should be used.

Static tests have been modified to account for possible differences in pyrite reactivity at coal mines. Caruccio et al., (1977) used reflected light microscopy to determine the morphology of pyrite which correlated with drainage quality from leaching tests. The fine grained "framboidal" pyrite was found to be the most reactive. Hammack (1986) used an evolved gas technique to provide a rapid determination of pyrite availability for oxidation. Samples were placed in a furnace, heated in an oxidizing atmosphere, and the evolved sulphur dioxide monitored at various temperatures. The pyrite oxidized from 150° C to 555° C, but only that which ignited below 220° C significantly affected drainage quality in parallel laboratory leaching tests.

Kinetic test methods attempt to model the acid producing and consuming processes in the laboratory or field. Various protocols have been developed as shown in Table 1. Kinetic tests might be expected to more accurately model the acid production and consumption processes though interpretation can be difficult. Kinetic tests are not commonly used because of their higher cost and longer duration, compared to static tests. B.C. Research has developed a type of kinetic test which assesses the ability of bacteria to oxidize sulphide minerals in a rock sample (Test 1 in Table 1). This test is of particular value in determining whether a mineral constituent of the waste is toxic to the microorganisms and inhibits their growth. Other kinetic tests follow the change in leachate quality over time and use the results in a physical or mathematical model of the AMD process. Mathematical models have recently been developed to quantitatively predict mine drainage quality (Caruccio, 1984; Jaynes et al., 1984; Ricca and Shultz, 1979; Halbert et al., 1983: Nicholson, 1984; Cathles, 1979; Davis and Ritchie, 1983). These models are rather complex and have not been extensively verified or used for prediction to date.

A major difficulty with prediction techniques is the lack of verification of test results with field experience. Two recent studies, one of metal mines and one of coal mines, examined the accuracy of some AMD prediction techniques.

Static (acid/base accounting) and kinetic (humidity cell) tests were conducted on 22 rock samples from seven metal mines in British Columbia and Yukon (Ferguson, 1985). Samples of tailings and waste rock were obtained from both active and abandoned minesites. The total sulfur contents ranged from 0.13 to 49.2 percent, and neutralization potentials ranged from 0 to 258 tonnes $CaCO_3$ equivalent per 1,000 tonnes of material. The very high percent sulfur contents of some samples yielded a definite prediction for acid production according to static methods. The static and kinetic tests correctly predicted the formation of AMD in all but six cases - three were incorrect and three were inconclusive. A few results from the kinetic tests (humidity cells) were difficult to interpret due to indefinite pH trends. The results supported the use of static tests for predicting AMD, but suggested there was only a poor relationship between the magnitude of the net base value (net neutralization potential) and the amount of acidity produced during the initial stage of the AMD process. The static tests may be useful for qualitative AMD predictions but not for estimates of actual mine water quality.

Correlations between the static test results (percent sulphur, neutralization potential, and net neutralization potential) and the kinetic test results (sulphate production rate, cumulative acidity production, initial leachate pH and final leachate pH) provided some insight into the geochemical processes involved in acid generation. As expected, the sulphate production rate increased with percent sulphur and was a good indicator of the sulphide oxidation rate. The initial pH of the leachate decreased with increased percent sulphur probably reflecting an increase in soluble metal salts in the samples at the start of the experiment. The cumulative acidity production and final leachate pH, however, correlated with the neutralization potential rather than the net neutralization potential as might be expected from static test prediction theory. In these tests, the quantity of neutralizing minerals in the samples appeared to play the critical role in initially controlling the acid generation process. In the longer term, acid production from sulphide minerals would be expected to "outlast" the alkalinity from carbonate minerals in potentially acid generating rocks and the AMD process would enter Stage II. These findings agree with results from other short-term (up to 6 months) kinetic tests where little acid was produced from samples exhibiting a strong acid potential according to static tests. Trends in sulphate production, acidity, and pH from kinetic tests must be examined together with the static test results to forecast the long term leachate quality at a mine.

An analogous study evaluated the predictive capability of static (acid-base account) and kinetic (a modified humidity cell procedure) tests applied to coal overburden (Erickson, 1986). Sites were deliberately selected to focus on hard-to-predict cases. In the preliminary analysis, 25 sites were analyzed by the static test; 14 of these were also subjected to the kinetic test.

A simple type of interpretation was applied to acid-base account data. The net neutralization potential (NNP) was calculated for each lithologic unit and the values for individual units were combined to generate a thickness-weighted average NNP to represent the whole overburden column. Thickness-weighted NNP values ranged from -20 to +250 tonnes $CaCO_3$ per 1,000 tonnes of overburden. These values were then compared to flow-weighted net alkalinity in drainage from reclaimed mine sections (Table 2). There is no linear correlation between observed drainage alkalinity and thickness-weighted NNP (Figure 5). NNP values greater than 33 tonnes/1,000 tonnes were always associated with alkaline drainage while NNP values less than 22 tonnes/1,000 tonnes were predominantly associated with acid drainage. The validity of a boundary between 22 and 33 tonnes/1,000 tonnes to distinguish sites that will not produce acidic drainage is uncertain because only 4 of 25 sites had higher NNP values. Further analyses are in progress to test other methods of evaluating overburden analysis data for water quality predictions.

Kinetic tests were conducted on overburden from 14 of these sites. The cumulative alkaline loads produced by each lithologic unit during the simulated weathering test were averaged for the whole overburden column on a thickness-weighted basis and compared to actual drainage quality. As with the acid-base account results, no linear correlation was found. Attempts to use the thickness-weighted alkaline load from the kinetic test to predict acid or alkaline drainage when the NNP value from the acid-base account was less than 22 tonnes/1,000 tonnes were unsuccessful.

These findings are not surprising in light of the high geochemical variability in coal overburden strata. Tests on individual lithologic units fail to account for interactions between potential acid- and alkaline-producing materials.

Uncertainty In AMD Production

The production of acid mine drainage is complex and yet, important factors can be modelled for prediction. However, uncertainty is always present and varies according to the nature of the waste material and its deposition and reclamation history. The range in uncertainty is illustrated in the four simple cases of waste rock disposal in Figure 6. Case one represents the most certain prediction where the waste is either completely composed (1a) or devoid (1b) of sulphide minerals; cases rarely seen at real mines. Uncertainty increases in case two where the sulphide content has increased or decreased from the two extremes in case one. The potential for AMD can be predicted with reasonable certainty by static tests in either high (2a) or low sulphide content (2c) samples. Predictions are more difficult when the sulphide and carbonate content are in critical balance (carbonates 3.125 times sulphides by weight) as shown in 2b. If every particle contains a balance of sulphide and carbonate minerals then theoretically, the two mineral types would be balanced. However, the maximum amount of pyrite that can be oxidized is unlimited, while carbonate dissolution is governed by equilibria. Further, coatings on both sulphide and carbonate mineral surfaces will form and affect the chemical reactions and ultimate drainage quality. Kinetic tests would be valuable in assessing reaction rates.

Homogeneous wastes are rarely found at real mines. More frequently, the acid generation potential will range reflecting spatial differences in geological environments. The numerical balance of the acid generation potential of the entire rock column would be assessed as part of the AMD prediction in acid/base accounting. This balance is important, but, the position of the waste types within the dump may ultimately determine drainage quality (Case 3). Potentially acid generating waste deposited on the exterior of a dump may generate AMD that may be only partially neutralized by carbonate minerals in the interior (3a). Coatings would form on carbonate minerals diminishing neutralization capability. In some cases, the sulphide minerals may be randomly distributed in the dump (3b). Acid generation may occur in only isolated zones and seep quality will vary depending upon its source. Potentially acid generating waste should be placed in the interior of the dump where only alkaline groundwater contacts the high sulphide waste (3c). This is AMD prevention by segregation. However, with large quantities, or highly reactive sulphides, the AMD may form despite the alkaline cover due to geochemical limitations on the maximum amount of alkalinity that can be delivered to the sulphide minerals, as discussed previously. A combination of static and kinetic tests, the latter using a simulated stacking sequence of wastes, might be valuable in assessing the relative rates of reaction.

Heterogeneous geochemistry, randomly distributed particles, and an overall sulphide/carbonate balance represents one of the most difficult predictions (Case 4). Very high uncertainty would be associated with a prediction in this case and must be supported by comprehensive kinetic tests on samples that simulate quantity and placement of various lithologic units. Case four describes the sites selected for the coal mine drainage prediction study described earlier. The inaccurate predictions identified in that study may demonstrate the role of spoil placement.

Determining the Source of AMD

The overall cost of AMD control would decrease if the most active areas within acid generating wastes could be identified and mitigation measures targeted for maximum benefit. Locating and sampling the seeps from mining wastes is usually the first step that companies and regulatory agencies take to identify acid generating zones. The pH, acidity, sulphate, metal, and microbial content, and flow of the seeps indicate the geochemical and hydrological processes within the waste. For example, sampling at an abandoned waste rock dump in British Columbia indicated that flow and quality of seeps varied, but their location remained the same; neutralization agents could be applied at these specific sites to reduce contaminant loading (Figure 7). The majority of copper released from the dump originated from two of the four drainage zones in the dump (zones BB and CC in Figure 7). Mitigation measures should be targeted to those areas.

At these sites, seasonal variation in AMD quantity and quality requires monitoring over a wide range of hydrologic conditions to quantify loadings to the receiving environment. Contaminated seeps are usually most readily identified in the spring and fall coincident with maximum melt or precipitation events following prolonged dry periods.

Yellow or red staining of rock and leachate may also indicate the source of AMD. AMD containing only ferrous iron is clear, but shows a red or yellow tint downstream as the ferrous iron is oxidized to the ferric form. AMD from metal mines may be blue-green reflecting high copper values. Dead vegetation near a seep may be an early indicator of contamination.

Characteristics of the waste such as total and sulphate sulphur, neutralization potential, and paste pH indicate the progress of AMD formation reactions. Samples of metal mine tailings can be obtained at various depths and analyzed by these geochemical tests to determine the depth of oxidation and other information useful in designing alkaline or impermeable covers for AMD control.

The partial pressure of oxygen available in the pore spaces throughout the waste indicates both oxygenated zones that can support acid production and zones where oxidation is occurring. Several oxygen profiles (O_2 partial pressure versus depth) are typical for mine wastes (Harris and Ritchie, 1985; Erickson, 1985). When oxygen is supplied by diffusion (induced by concentration differential) from the atmosphere, the profile shows decreasing oxygen with increasing depth. Active oxidation zones are reflected as sharp partial pressure gradients. When advection (induced by thermal or pressure differential) supplies oxygen to the mine waste, high partial pressures are observed at intermediate depths. Gradients are less easily correlated with active oxidation in these cases. However, the availability of oxygen in advection zones indicates that, if present, pyritic material will likely undergo oxidation.

Variations in the surface or interior temperatures of mining wastes can indicate acid generation since the reactions are exothermic. Temperatures up to 45° C were recorded by thermister strings buried in a very active acid generating waste rock dump in British Columbia (Patterson, 1986). A significant drop in temperature following reclamation of a waste rock dump was recorded by Harries and Ritchie (1986) using thermister probes in wells. Thermography (infrared imaging) was used to identify acid generating zones at three waste rock dumps and open pits in British Columbia. A related, but simpler approach, involved locating bare spots on a waste rock dump following the first snowfall of winter. In all cases, heat generation correlated with observed differences in water quality of seeps.

Remote detection of acid sources is possible with the geophysical method of electomagnetic induction measurements. This means of surveying detects terrain conductivity anomolies. In several surveys conducted at regraded coal mines, anomolies were observed and later confirmed to reflect pyrite oxidation, changes in saturation state, and disturbed area boundaries (Watzlaf and Ladwig, 1987). The method was found to be most useful as a reconnaissance tool preceding application of detailed monitoring methods.

Summary

Though the acid mine drainage process is complex and not completely understood, prediction techniques are available that are successful for many mining projects. Considerable skill and experience is required to conduct some tests and interpret results for accurate forecasts of mine drainage quality. Uncertainity is always present, and should be estimated and acknowledged in the prediction statement.

The location of acid generating zones within a mining waste can be determined by both simple and sophisticated techniques. In general, it is easier to locate the source of AMD than predict prior to mining if drainage will be acidic. Success in both of these areas of prediction are required if mining is to be conducted in an environmentally sound manner.

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TABLE 1. - KINETIC ACID MINE DRAINAGE PREDICTION TECHNIQUES

(Adapted from Ferguson, 1985 and Ferguson and Erickson, 1986)

TEST	DESCRIPTION	USE	REFERENCES
1. B.C. Research Confirmation Test	- Sample placed in 250 ml Erlenmeyer flask with 70 ml nutrient media, culture of Thiobacillus ferrooxidans at pH 2.2-2.5. - Flask placed on gyratory shaker at 35° C in CO_2 enriched atmopshere. - pH monitored and additional sample added. - If pH rises substantially then sample non-acid producer. - If pH remains low then sample potential acid producer.	Frequently used for metal mines in Canada	Bruynesteyn and Hackl (1984)
2. Shake Flasks	- Sample placed in 1L Erlenmeyer flask with 600 ml of water or nutrient solution. - Series of samples tested at various starting pH, inoculation, and temperature. - Samples incubated for up to 3 months. - Leachates analyzed weekly and bi-weekly for range of parameters.	Infrequently used for metal mines in Canada	Davidge (1984) Halbert, et al., (1983)
3. Soxhlet Reactor	- Standard or modified Soxhlet reactor used. - Water placed in reservoir, vaporized and passed into condenser. Condensed liquid drips into thimble with sample and thence back into the reservoir. - Leachate analyzed after 64-192 hrs. for a range of parameters.	Infrequently used in coal fields of Appalachia, U.S.A.	Sullivan and Sobek (1982) Renton (1983)
4. Humidity Cell	- Sample placed in plexiglass container connected to humidified air. - Humidified air passed over sample for 3 days. - Dry air passed over sample for 3 days and 200 ml of water added on seventh day. - Leachate removed and analyzed for range of parameters. - Procedure repeated for 8-10 weeks.	Relatively common in coal fields of Appalachia, U.S.A.	Caruccio et al., (1977) Caruccio et al., (1980)

TABLE 1. - KINETIC ACID MINE DRAINAGE PREDICTION TECHNIQUES (Continued)

(Adapted from Ferguson, 1985 and Ferguson and Erickson, 1986)

TEST	DESCRIPTION	USE	REFERENCES
5. Columns/ Lysimeters	- Sample placed in column and periodically leached by distilled water. - Samples of leachate analyzed for range of parameters. - Usually leached for 8-10 weeks minimum. - Several variations of set-up and leaching procedures in literature.	Relatively common in Canada and U.S.A.	Sturey et al., (1982) Apel (1983) Ritcey and Silver (1981)
6. Test Plots/ Pits/Piles	- Run of mine or modified sample placed on impervious surface. - Precipitation provides leachate which is collected in sump. - Samples of collected leachate analyzed for range of parameters. - Test usually run for at least 1 year.	Very infrequently used in Canada and U.S.A.	Eger et al., (1981) Murray and Okuhara (1980)

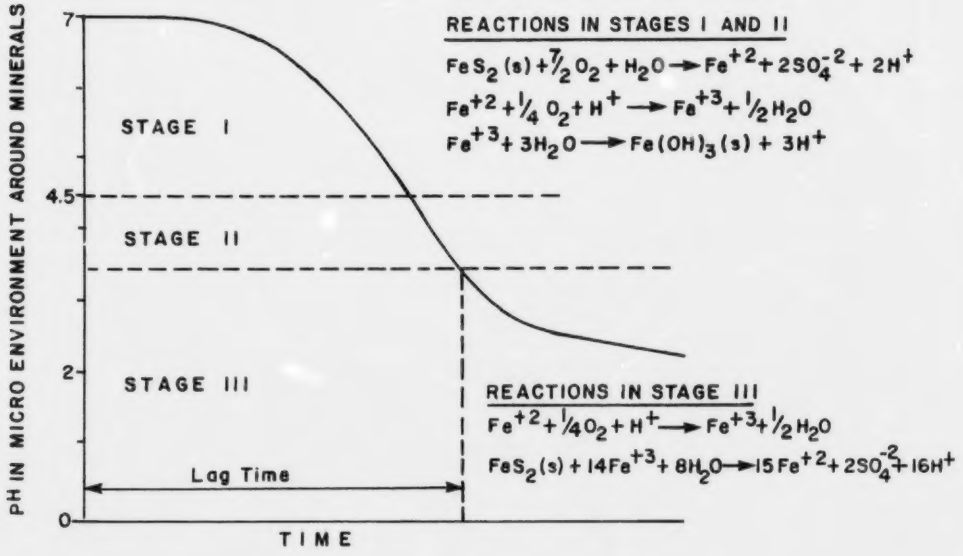
TABLE 2. - COMPARISON OF OVERBURDEN ANALYSIS AND POST-MINING DRAINAGE QUALITY FOR 25 U.S. COAL MINES(1)

SITE	ACID-BASE ACCOUNT Thickness-weighted net neutralization potential, tonnes/ 1,000 tonnes as $CaCO_3$.	MODIFIED HUMIDITY CELL Thickness-weighted cumulative net alkaline load, g/300g as $CaCO_3$.	POST-MINING DRAINAGE QUALITY Flow-weighted net alkalinity, mg/L as $CaCO_3$.
1	-1.6	--(2)	-120.
2	-11.	--	7.6
3	7.5	--	65.
4	4.8	18.	-72.
5	250.	18.	170.
6	34.	33.	110.
7	-20.	--	-340.
8	-2.9	-18.	-110.
9	-3.5	--	-85.
10	-2.9	--	-100.
11	-6.8	-170.	100.
12	3.0	2.2	-290.
13	7.4	23.	110.
14	3.2	7.7	-860.
15	22.	--	-13.
16	11.	--	-240.
17	150.	--	100.
18	-4.6	-210.	-500.
19	2.7	-24.	-190.
21	3.2	27.	-8.7
22	90.	-280.	430.
25	20.	21.	310.
27	9.8	--	52.
29	3.6	-42.	-610.
30	-0.49	--	38.

(1) All values have been rounded to 2 significant figures.

(2) Dashes indicate that the kinetic test was not performed.

STAGES IN THE FORMATION OF ACID MINE DRAINAGE



REACTIONS IN STAGES I AND II

$FeS_2(s) + \frac{7}{2}O_2 + H_2O \rightarrow Fe^{+2} + 2SO_4^{-2} + 2H^+$

$Fe^{+2} + \frac{1}{4}O_2 + H^+ \rightarrow Fe^{+3} + \frac{1}{2}H_2O$

$Fe^{+3} + 3H_2O \rightarrow Fe(OH)_3(s) + 3H^+$

REACTIONS IN STAGE III

$Fe^{+2} + \frac{1}{4}O_2 + H^+ \rightarrow Fe^{+3} + \frac{1}{2}H_2O$

$FeS_2(s) + 14Fe^{+3} + 8H_2O \rightarrow 15Fe^{+2} + 2SO_4^{-2} + 16H^+$

OVERALL PROCESS (Stumm and Morgan, 1981)

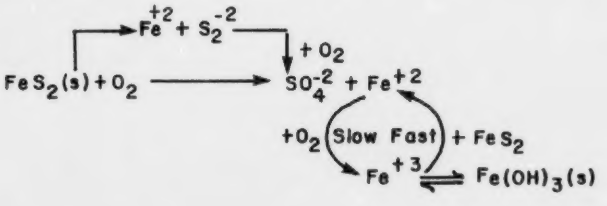

$Fe^{+2} + S_2^{-2}$
$+ O_2$
$FeS_2(s) + O_2$
$SO_4^{-2} + Fe^{+2}$
$+O_2$
Slow
Fast
$+ FeS_2$
Fe^{+3}
$Fe(OH)_3(s)$

FIGURE 1 FORMATION OF ACID MINE DRAINAGE FROM PYRITE OXIDATION

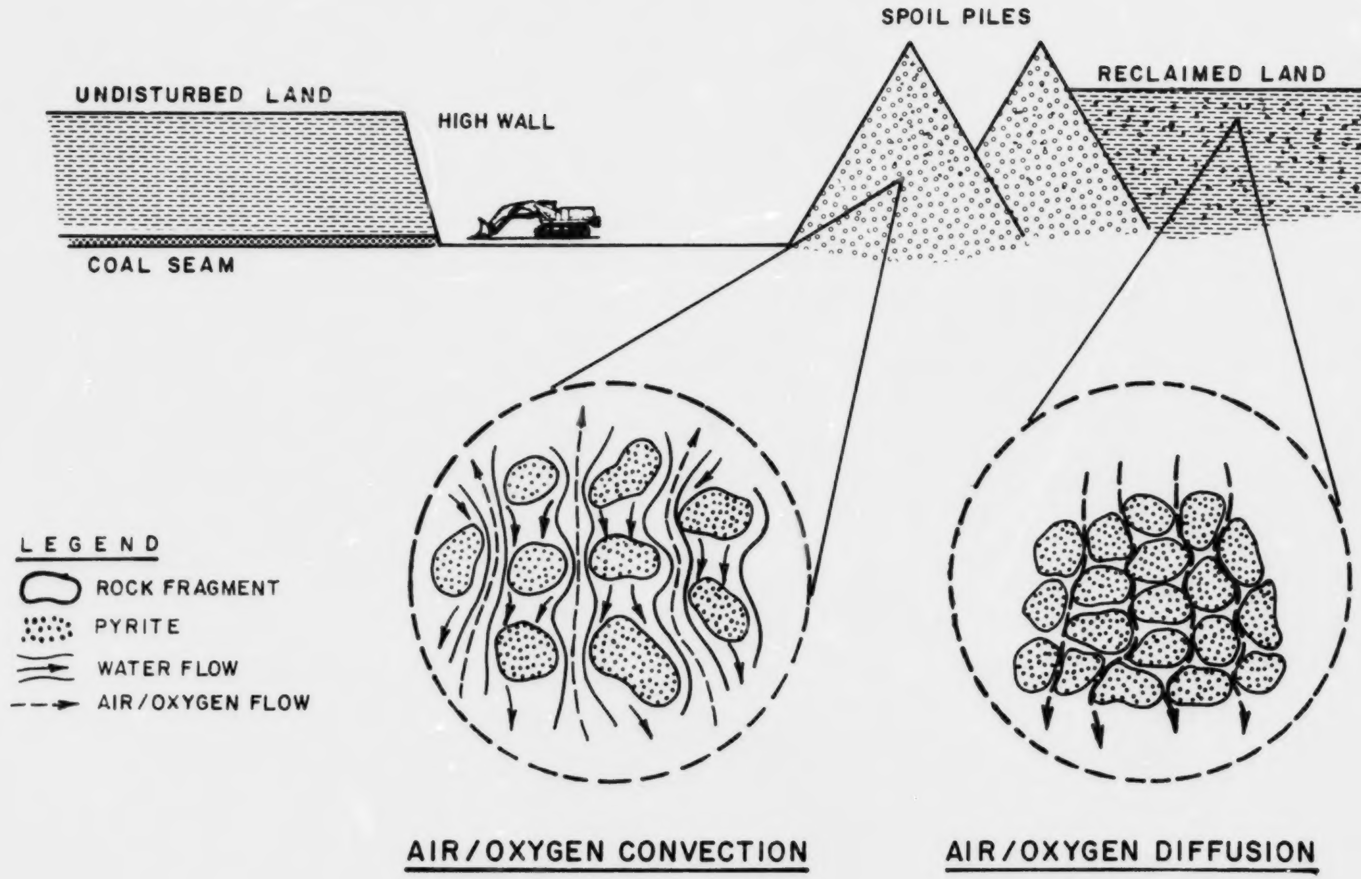
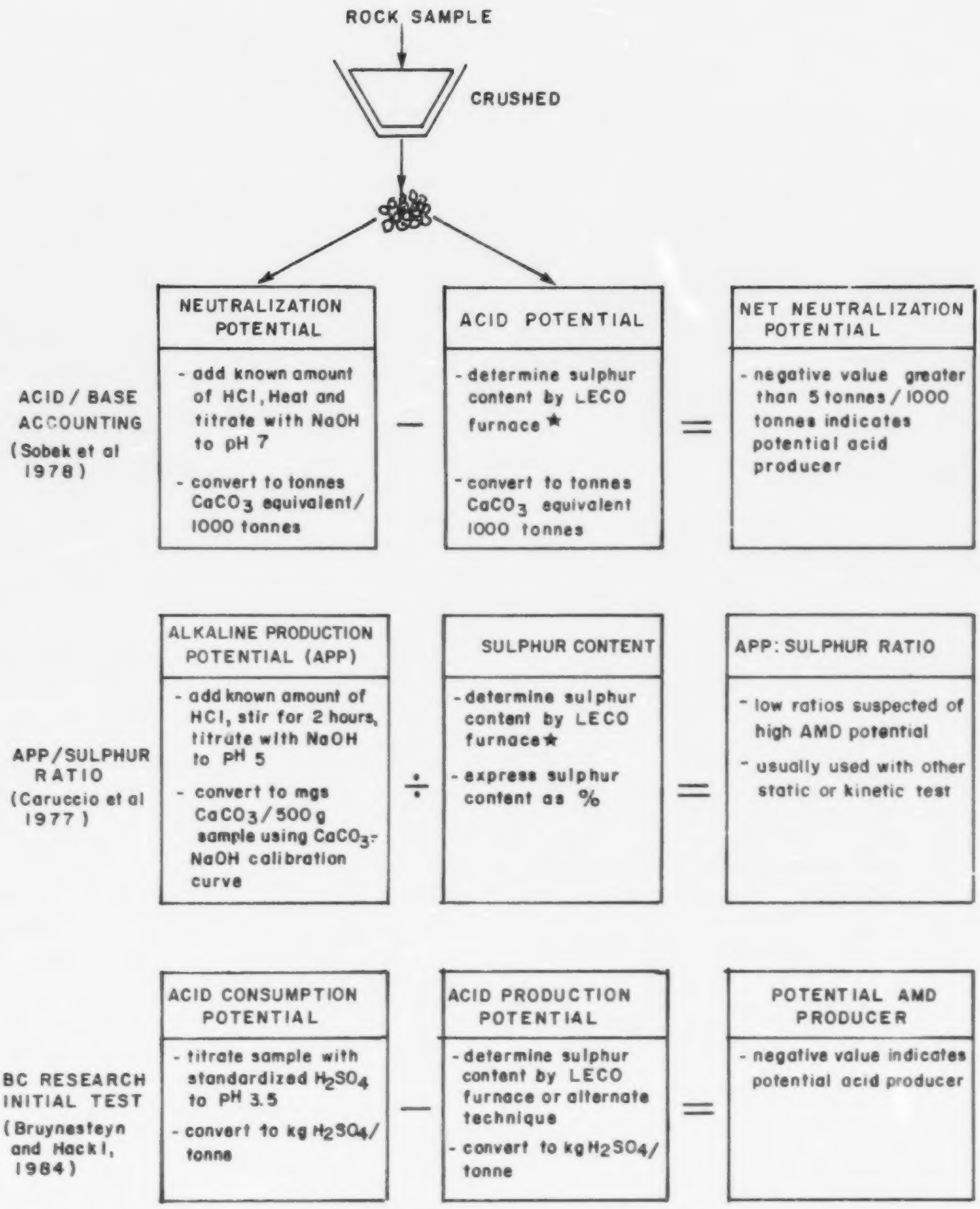

SPOIL PILES
UNDISTURBED LAND
HIGH WALL
RECLAIMED LAND
COAL SEAM
LEGEND
ROCK FRAGMENT
PYRITE
WATER FLOW
AIR/OXYGEN FLOW
AIR/OXYGEN CONVECTION
AIR/OXYGEN DIFFUSION

FIGURE 2 PRODUCTION OF ACID MINE DRAINAGE IN POROUS WASTE DUMPS (AFTER CATHLES 1982)



ACID / BASE ACCOUNTING (Sobek et al 1978)	**NEUTRALIZATION POTENTIAL** - add known amount of HCl, Heat and titrate with NaOH to pH 7 - convert to tonnes $CaCO_3$ equivalent/ 1000 tonnes	−	**ACID POTENTIAL** - determine sulphur content by LECO furnace ★ - convert to tonnes $CaCO_3$ equivalent 1000 tonnes	=	**NET NEUTRALIZATION POTENTIAL** - negative value greater than 5 tonnes / 1000 tonnes indicates potential acid producer
APP/SULPHUR RATIO (Caruccio et al 1977)	**ALKALINE PRODUCTION POTENTIAL (APP)** - add known amount of HCl, stir for 2 hours, titrate with NaOH to pH 5 - convert to mgs $CaCO_3$/500 g sample using $CaCO_3$-NaOH calibration curve	÷	**SULPHUR CONTENT** - determine sulphur content by LECO furnace★ - express sulphur content as %	=	**APP: SULPHUR RATIO** - low ratios suspected of high AMD potential - usually used with other static or kinetic test
BC RESEARCH INITIAL TEST (Bruynesteyn and Hackl, 1984)	**ACID CONSUMPTION POTENTIAL** - titrate sample with standardized H_2SO_4 to pH 3.5 - convert to kg H_2SO_4/ tonne	−	**ACID PRODUCTION POTENTIAL** - determine sulphur content by LECO furnace or alternate technique - convert to kg H_2SO_4/ tonne	=	**POTENTIAL AMD PRODUCER** - negative value indicates potential acid producer

★ Pyrite sulphur content may be determined for coal mine samples

FIGURE 3 STATIC ACID MINE DRAINAGE PREDICTION TECHNIQUES

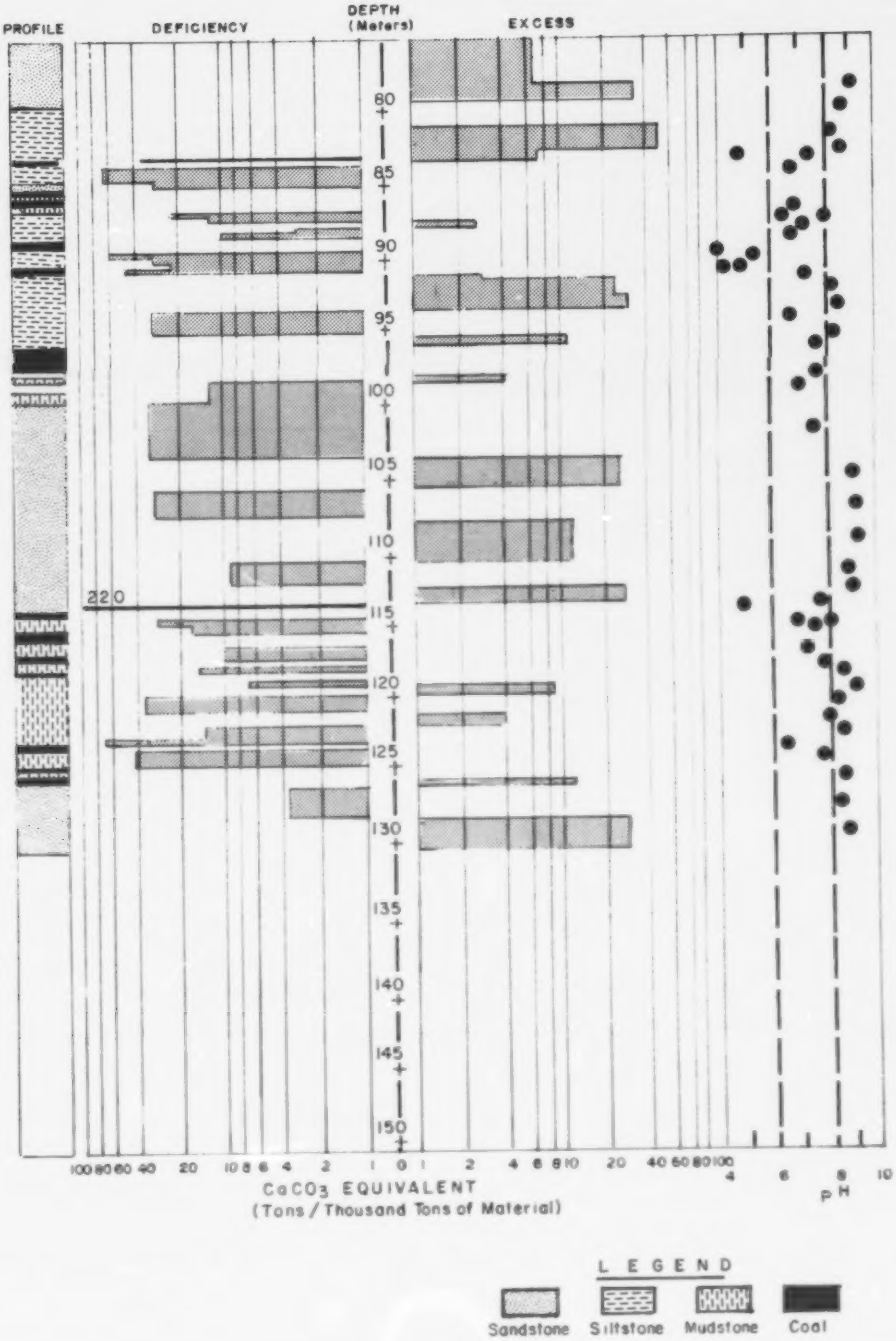


FIGURE 4   ACID/BASE ACCOUNT FOR B.C. COAL MINE
(FROM STURM ENVIRONMENTAL SERVICES, 1985)

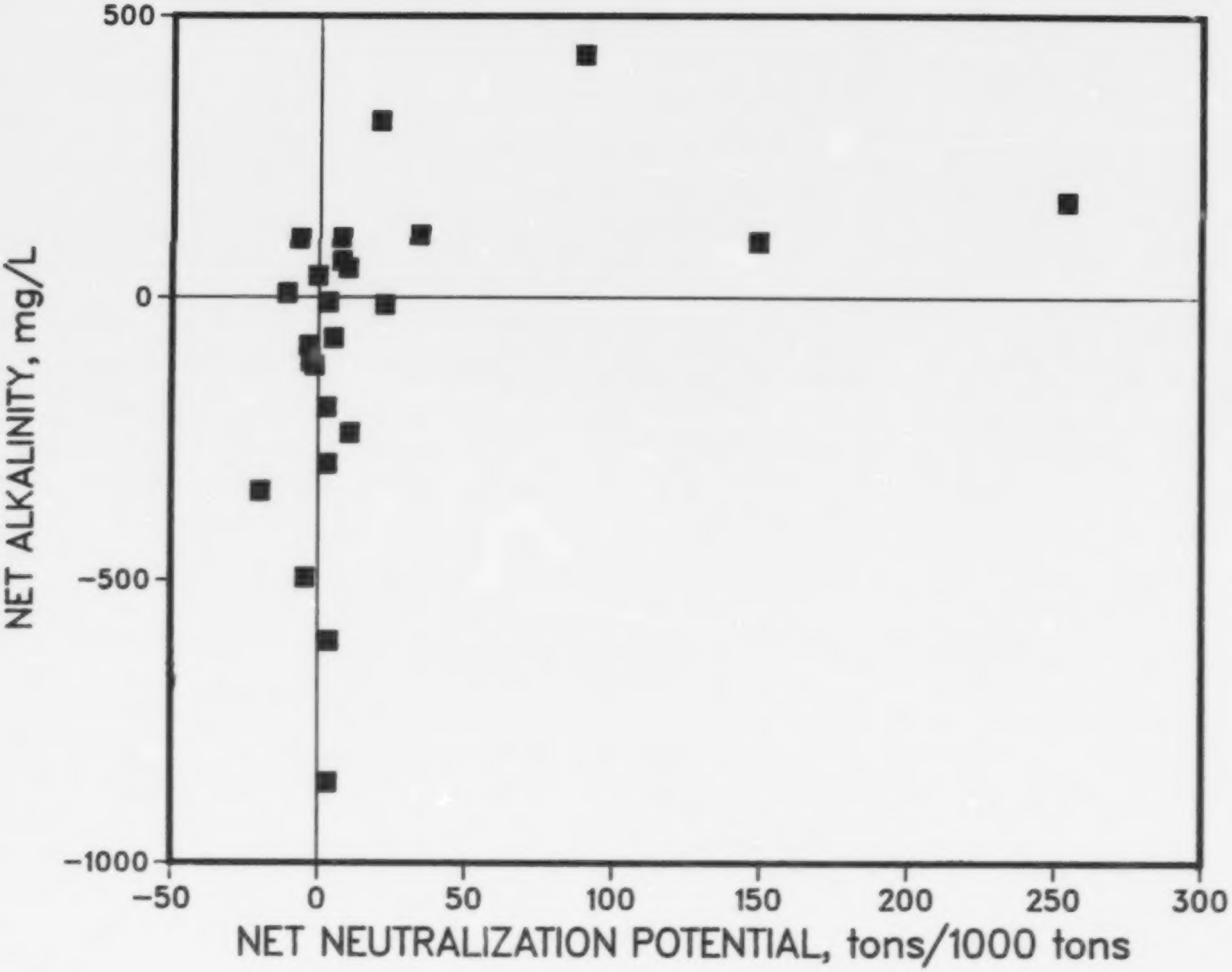


FIGURE 5 RELATIONSHIP BETWEEN POST-MINING WATER QUALITY (NET ALKALINITY) AND ACID-BASE ACCOUNT PREDICTOR (THICKNESS-WEIGHTED NET NEUTRALIZATION POTENTIAL)

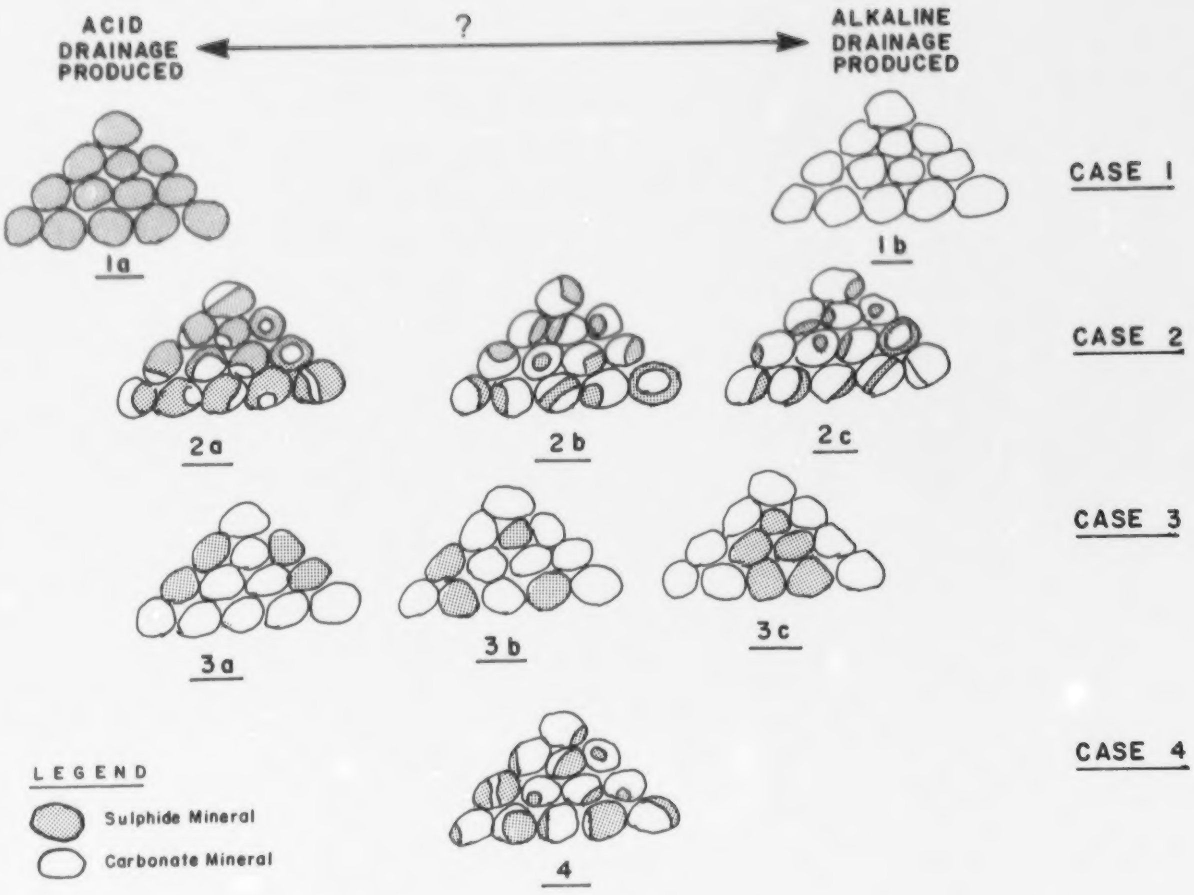

ACID DRAINAGE PRODUCED
?
ALKALINE DRAINAGE PRODUCED
1a
1b
CASE 1
2a
2b
2c
CASE 2
3a
3b
3c
CASE 3
4
CASE 4
LEGEND
Sulphide Mineral
Carbonate Mineral

FIGURE 6 UNCERTAINTY IN AMD PREDICTION

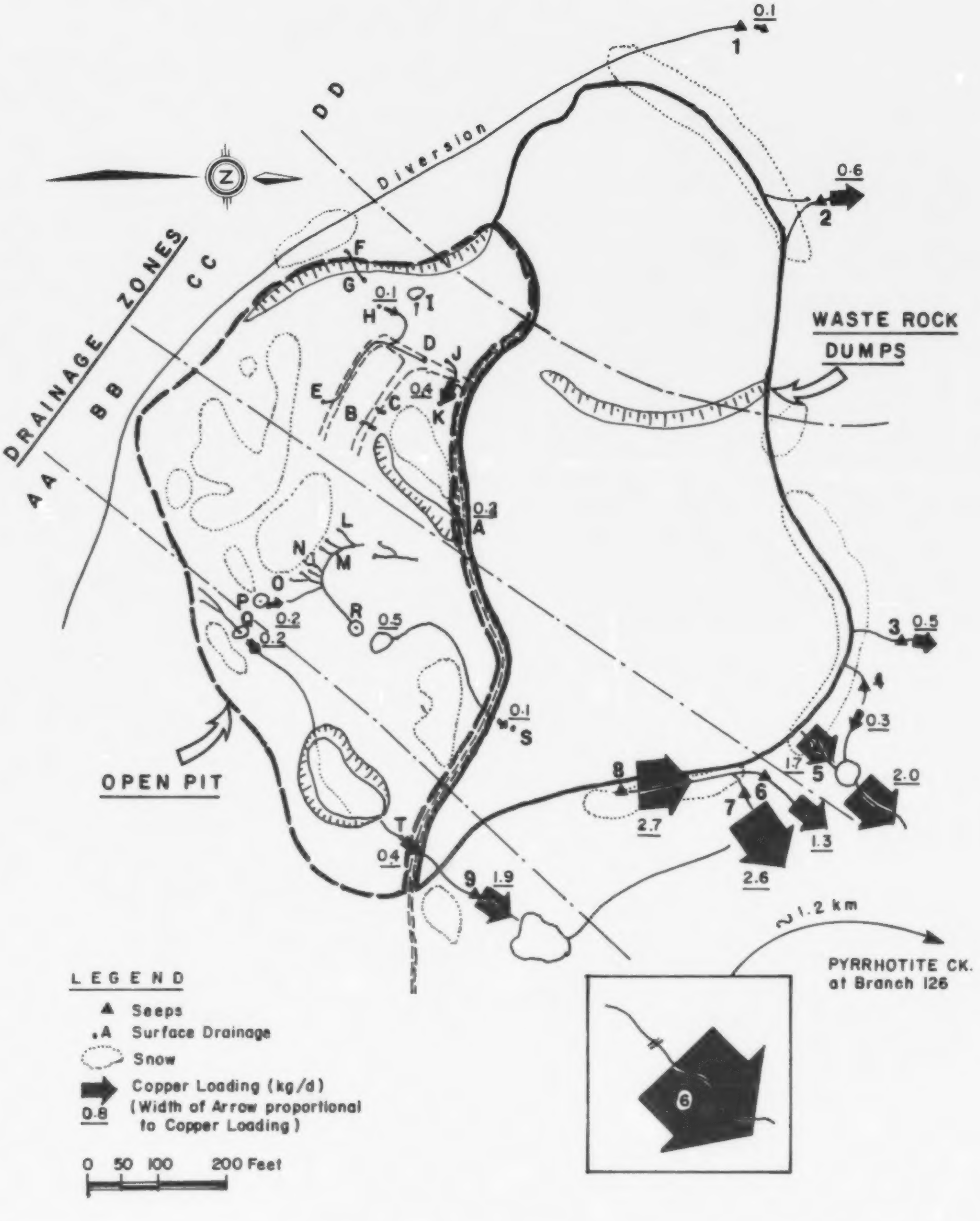

DRAINAGE ZONES
AA
BB
CC
DD
Diversion
WASTE ROCK DUMPS
OPEN PIT
0.1
1
0.6
2
3
0.5
4
0.3
1.7
5
2.0
8
2.7
7
6
2.6
1.3
9
1.9
T
0.4
S
0.1
F
G
0.1
H
I
D
J
E
B
C
0.4
K
0.3
A
L
N
M
O
P
Q
0.2
0.2
R
0.5
~1.2 km
PYRRHOTITE CK. at Branch 126
6
LEGEND
Seeps
A Surface Drainage
Snow
Copper Loading (kg/d)
0.8 (Width of Arrow proportional to Copper Loading)
0 50 100 200 Feet

FIGURE 7 METAL MINE OPEN PIT AND WASTE ROCK DUMP - SURFACE DRAINAGE AND SEEPAGE COPPER LOADINGS

PREDICTION OF ACID DRAINAGE IN MEGUMA SLATES

RODGER ALBRIGHT

ENVIRONMENT CANADA, DARTMOUTH, N.S.

ABSTRACT

Since 1984, evaluation and testing procedures for predicting acid drainage from Meguma slates in Nova Scotia have received close scrutiny due to inconsistencies observed between predictions based on lab analysis and field conditions. Preliminary studies suggest that one of the two commonly used tests for acid consuming potential may not be appropriate for field conditions.

The emphasis of the report is to reinforce the concept that prediction methods and their interpretation must be field applicable. Failure to field test and to know the limitations of such predictive tools can have very damaging results on the environment.

Several years ago I visited a school principal and noticed that for the benefit of all visitors, he had displayed the sign "If you think education is expensive, try ignorance."

This concept can easily be transferred to the topic of this section of the seminar "If you think it is time consuming and costly to predict impacts from slate disturbances, try the environmental and financial cost of ignorance." The degradation of 3 river systems adjacent to Hwy 102 and the Halifax International Airport (HIA) in the 60s and 70s resulted from such ignorance (Figure 1).

HALIFAX INTERNATIONAL AIRPORT & ADJACENT LANDS

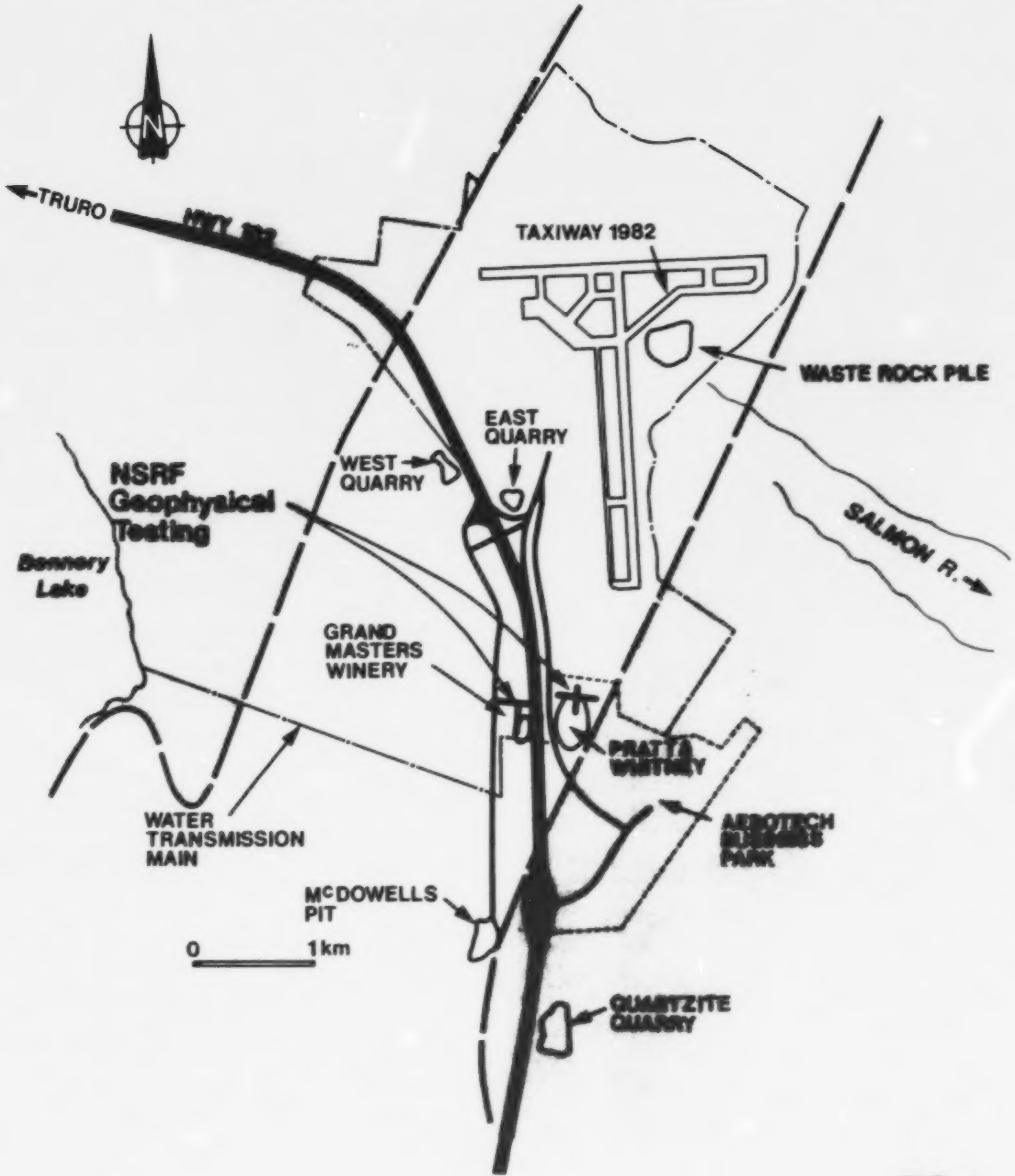

N
TRURO
HWY 102
TAXIWAY 1982
WASTE ROCK PILE
EAST QUARRY
WEST QUARRY
NSRF Geophysical Testing
SALMON R.
Bennery Lake
GRAND MASTERS WINERY
PRATT & WHITNEY
WATER TRANSMISSION MAIN
AEROTECH BUSINESS PARK
McDOWELLS PIT
0
1km
QUARTZITE QUARRY

FIG. 1

More recently, the significant degradation of the Salmon River Watershed resulting from the construction of taxiway Delta at HIA in 1982, coupled with nearby other proposals such as development of Aerotech Business Park made Environment Canada staff realize that although the situation in 1983 was certainly undesirable, the situation would steadily deteriorate if we did not initiate strong corrective action. It would be necessary to find out better methods to adequately predict, assess and control the problem of acid drainage from meguma slates.

About mid-1984, Environment Canada contracted the Nova Scotia Research Foundation (NSRF) to conduct a preliminary evaluation of some geophysical techniques for locating and distinguishing zones of mineralization in the slate formations. Two sites adjacent HIA were chosen (Fig. 1). Of the instruments used to test optimal methods for the detection of sulphides in the Halifax slates there was a good correlation between the anomalies located by both the electromagnetic ground conductivity and VLF EM units. Further, ground truthing suggested positive correlations between increased ground conductivity measurements and the presence of significant quantities of sulphides (largely iron sulphide) in the bedrock. By comparison, there was little, or no correlation between the magnetic anomalies and the ground conductivity anomalies.

Equally important, ground truthing associated with the study indicated strong variability along transects, between adjacent duplicate samples, and in samples differing in depth (Figures 2 and 3). Thus each sample analysis is applicable only to that sample, and results may vary significantly 1 m or 20 m away.

Also in 1984, the infrastructure for Aerotech Business Park was being designed under the policy of:

> **Minimal mineralized slate disturbance in order to prevent conditions conducive to acid water generation.**
>
> Building on or near slates requires investigation of alternate design and construction techniques which will not necessitate disturbance of mineralized slates. Where mineralized slate disturbance is considered necessary, and is approved, appropriate mitigation measures, treatment facilities and monitoring may be required both at the area of excavation and the slate disposal area. Environmental agencies acknowledge that applying alternate designs, use of mitigation measures, and monitoring programs may significantly increase costs of site development. The caution is expressed that use of mitigation measures will not guarantee projects without impacts related to acid drainage.

CHEMICAL ANALYSIS FOR ROCK SAMPLES FROM THE AEROTECH PARK GRID

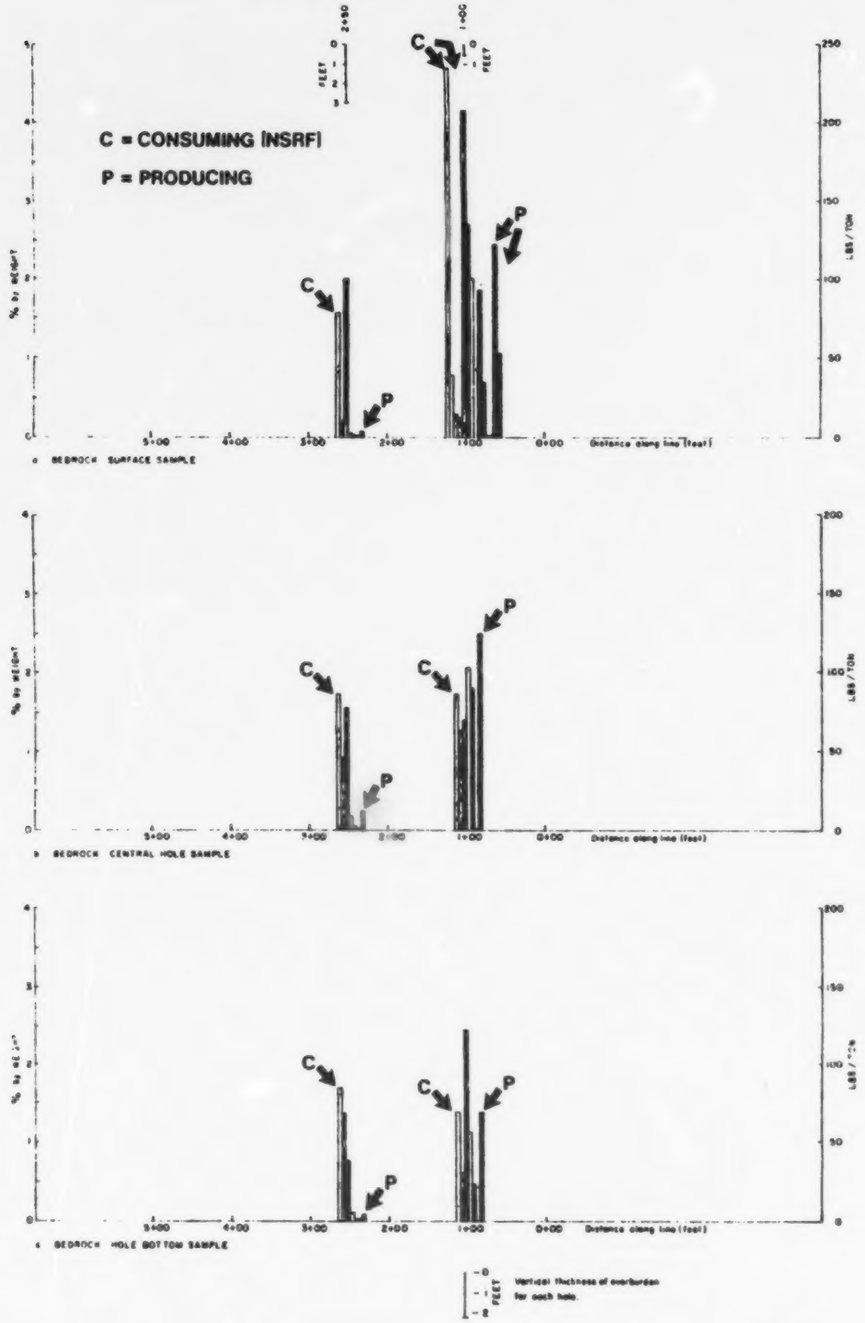


FIG. 2

CHEMICAL ANALYSIS FOR ROCK SAMPLES FROM THE GRAND MASTERS WINES GRID

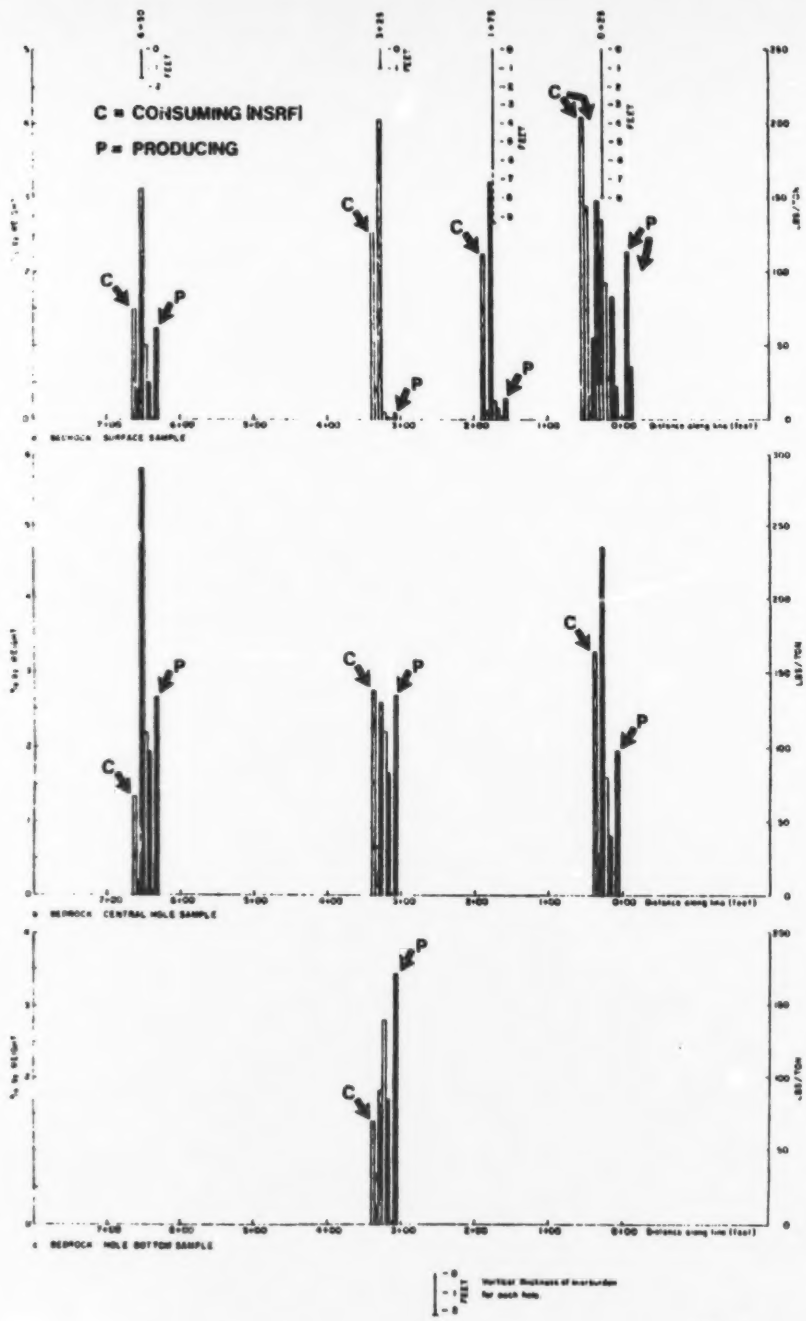


FIG. 3

This policy remains in effect today.

To use Aerotech Business Park as an example, for the 3.8 km water transmission main from Bennery Lake, approximately 3.0 km was built above slate within a clay berm. On-site roads were built without disturbing slate, although slate was exposed. The road subgrade, often an "Approved" slate from local quarries, was replaced by quartzite. At the same time, very conservative designs were applied to the proposed Pratt and Whitney and Grand Masters Wine Facilities (Figure 1).

Here it is important to note the common use of slates as road or other fill material from the 60-80 known slate quarries in N.S.. Use of mineralized slate can create environmental problems at the infill area, as well as at the quarry.

The real problem with the predictive ability of the assessment process and testing procedures in place at that time was highlighted in late 1985 when a contractor applied to use an existing slate quarry (McDowells pit) for construction of the proposed Aerotech Park Interchange on Hwy 102 (Figure 1). Approximately 170,000 m^3 of material was required for the interchange. As this volume was comparable to the volume disturbed for the HIA taxiway construction in 1982, Environment Canada, as DRIE advisor on environmental matters, took a very active interest in this proposal.

Under a typical Nova Scotia Quarrying Permit, a single but "representative" composite sample would be taken by the contractor and sent for analysis. The McDowell pit material was investigated for the acid production/ consumption ratio using the back titration method (NSRF unpublished), and the contractors sample was determined by provincial agencies to be environmentally acceptable. However, visual zones of pyrite mineralization were noted and spot sampling by EPS of standing water in the quarry indicated pHs in the range of 2.9 - 3.2, indicative of acid generation. At that time Environment Canada made a strong case against the use of the slate quarry. At the request of NSDOE and EPS, the contractor agreed to take extra samples which suggested weathered surface samples would pass, but that deeper unweathered samples would fail. Based upon the multiple samples, NSDOE refused permission to use McDowells Pit for the infill for the Aerotech Park Interchange.

Several obvious questions and inconsistencies were apparent after this quarry review.

1) The theory of the acid test is basically one of mass balance (i.e. for every unit of acidity, a unit of alkalinity would be immediately available to neutralize it) so on the whole, no acid generation should occur. The possibility that differing reaction rates could result in most of the acid being generated faster than

consuming ability is not considered.

2) The variation in acceptability of the several samples taken at the quarry in question clearly demonstrated the problem of 1 sample analysis for 170,000 m^3 of slate excavation. Previous samples taken and analyzed by NSRF in the geophysical study mentioned above confirmed variability of adjacent samples.

3) The standing water samples conflicted with the acid production/consumption test predictions. Further sampling of standing waters in other approved sites confirmed this conflict.

4) In reviewing a study by Mark King for IWD (1985) it was observed that consistently lower values of acid consumption were reported in samples taken in Lunenburg area than were reported for N.S. Department of Transportation (NSDOT) quarry tests for the same area. It was noted that IWD had samples analyzed after Duncan 1972, whereas NSDOT had samples analyzed using the back titration method.

5) The size of the particle as analyzed in the Lab (100 mesh) may substantially alter the availability of either acid or consuming abilities compared to the larger particles (0.5 - 10 cm) exposed under natural

conditions. The variation should be identified.

While Environment Canada has some concerns with the sampling and evaluation process, the major concern related to confidence in the lab tests. Environment Canada was reasonably confident in abilities of testing to predict acid generation potential, but queried why the reported acid consuming properties were not neutralizing the acidity, as was predicted. In late 1985, NSRF was contracted to do some preliminary investigations on the acid consuming potential of Halifax formation slates.

A report "Mineral Composition and Acid Consuming Potential of Nova Scotia Shales" was produced, and although I don't understand all the theory behind some aspects of the study, I would like to spend some time on its discussions and results.

The acid consuming potential of rock is a measure of the rocks ability to remove hydrogen ions from solution thus raising or maintaining the pH. Acid consuming potential is reported as the number of kilograms of sulphuric acid consumed by a tonne (MG) of sample (kg H_2SO_4/MG). If the acid consuming potential of a rock sample exceeds the acid production potential then the rock should not be a source of acid water. (Assumes mass balance theory, see item 1.).

The two acid consuming tests used in the evaluation are described briefly below, and illustated schematically in Figure 4.

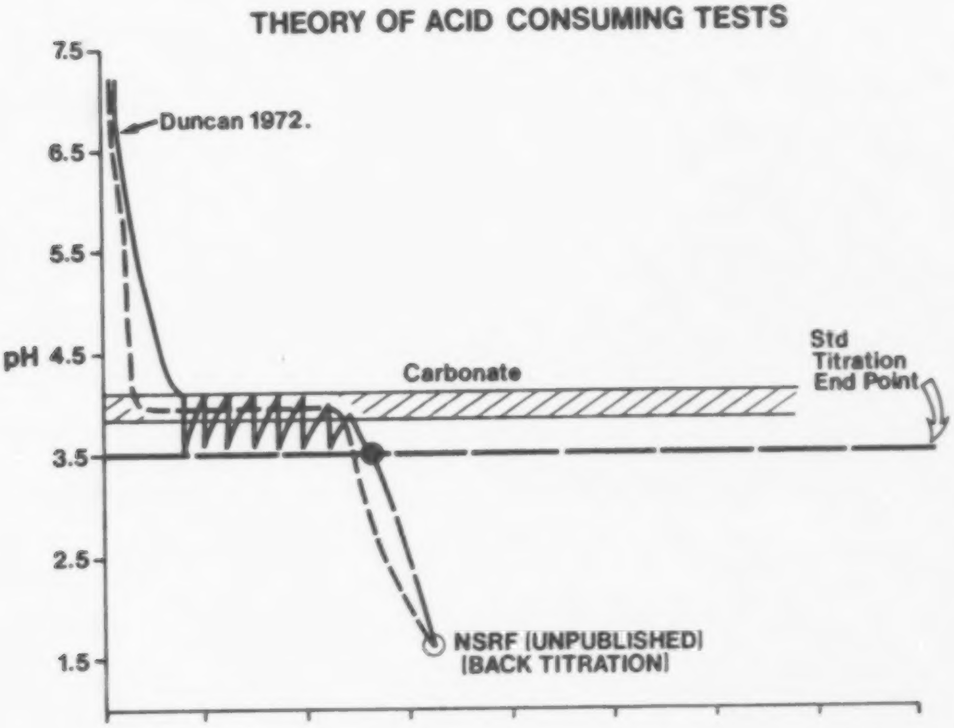


FIG. 4

1) Duncan - (B.C. Research, Project No. 1499, 1972) - titrate a sample to a pH of 3.5, stir and continue titration when sample changes pH by 0.1 unit within a 4 hour time period.

2) Back titration of excess acid - (Nova Scotia Research Corporation, Unpublished) - thirty mls of 0.1N H_2SO_4 is added to 10 g of sample. Reaction is allowed to

proceed and the excess acid titrated to a pH of 3.5. The amount of acid consumed by the sample provided the acid consuming potential. (Note: The pH of the sample plus solution before back titration is normally in the 1-2 range.)

Analysis of various mesh size fractions are presented in Tables 1, 2 and 3. As the size of the particles decrease, there is an increase in the common cation exchange capacity, and apparent alkalinity. This suggests that the increase in acid consuming power is due to reactions at the surface of particles which increases with decreasing particle size. Comparison of common cation exchange capacity and alkalinity shows that common cation exchange accounts for only 0.1% of the alkalinity and will be ignored in future discussions.

The apparent alkalinities show variations from one pit to another and in general an increase in value as the particle size decreases. Comparison of values show that carbonate does not account for the total alkalinity of the slates, but only for amounts varying between 1-50%. There is at least one other component present contributing 50-99% of the alkalinity. The presence of a third component is substantiated by the presence of alkalinity in the samples from the Buckley Lake Pit and the Pratt and Whitney Pit in which very little carbonate was detected but showed significant alkalinity.

TABLE 1

Cation Exchange Capacity

Values Recorded as lbs H_2SO_4/Ton of Sample

	Rock Quarry*				
Mesh Size	1	2	3	4	5
-4 + 8	4.3×10^{-3}	3.2×10^{-3}	5.0×10^{-3}	2.3×10^{-3}	4.5×10^{-3}
-8 + 16	7.7×10^{-3}	4.8×10^{-3}	6.5×10^{-3}	2.9×10^{-3}	8.8×10^{-3}
-16 + 32	1.1×10^{-2}	5.8×10^{-3}	1.2×10^{-2}	5.1×10^{-3}	1.3×10^{-2}
-32 + 50	1.2×10^{-2}	8.8×10^{-3}	1.1×10^{-2}	6.9×10^{-3}	1.5×10^{-2}
-50 + 100	1.4×10^{-2}		1.1×10^{-2}	9.0×10^{-3}	1.6×10^{-2}
-100 + 200	1.4×10^{-2}	1.1×10^{-2}	1.2×10^{-2}	9.5×10^{-3}	1.5×10^{-2}

TABLE 2

Available Carbonates As Sulphuric Acid

Percent Carbonates Determined and Expressed as lbs H_2SO_4/Ton of Sample

Mesh Size	1	2	3	4	5
-4 + 8	L 1.63	11.4	4.9	1.63	4.9
-8 + 16	L 1.63	L 1.63	11.4	3.3	1.6
-16 + 32	L 1.63	L 1.63	3.3	4.9	1.6
-32 + 50	L 1.63	L 1.63	4.9	4.9	8.2
-50 + 100	L 1.63	L 1.63	1.63	L 1.63	9.8
-100 + 200	L 1.63	L 1.63	6.5	L 1.63	7.5

TABLE 3

Apparent Alkalinity

Values Expressed as lbs H_2SO_4/Ton of Sample

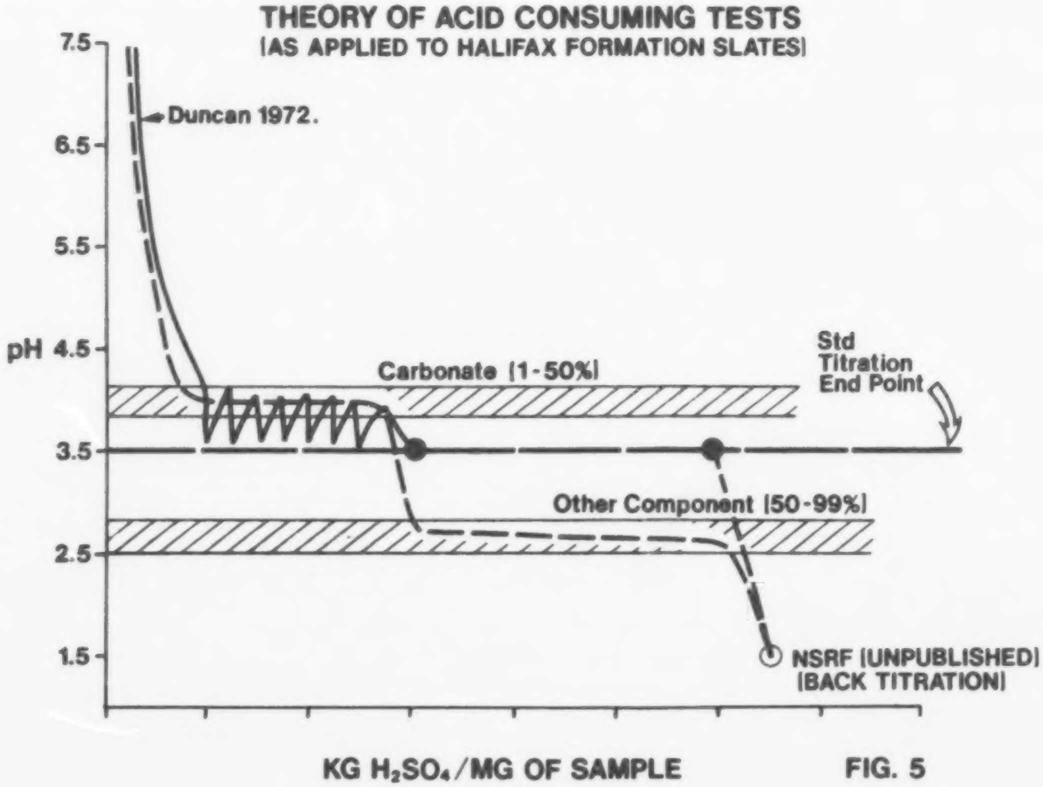
Mesh Size	1	2	3	4	5
-4 + 8	6.6	7.1	8.5	12.2	12.4
-8 + 16	8.1	8.7	7.8	14.8	12.7
-16 + 32	13.9	10.7	12.2	17.3	17.3
-32 + 50	22.7	20.2	37.7	28.8	37.6
-50 + 100	26.9	25.0	44.1	59.5	45.6
-100 + 200	32.1	55.4	40.9	47.4	41.2

* 1. Buckley Lake Pit
2. Pratt & Whitney
3. Water Transmission Main - Highway 102
4. Water Transmission Main - Airport Hotel Road
5. McDowell Pit

L = less than

Samples of slate from the various pits and mesh fractions were titrated with sulphuric acid. It was thought that hydrogen ion absorbing components, other than common cation exchange capacity and carbonate, may be detected by plotting titration curves.

The occurrence of inflection points in the acid titration curves and the disappearance of X-Ray diffraction lines with acid treatment suggests that additional alkalinity by some other component is present in slates in the pH range of 2.5 to 2.8. If these reactions do not occur or are very slow at pHs greater than 3.5, then the acid consuming potential determined by back titration will be high (Figure 5).



THEORY OF ACID CONSUMING TESTS
(AS APPLIED TO HALIFAX FORMATION SLATES)

FIG. 5

In such a test, the sample is treated with acids stronger than commonly observed in nature, promoting acid-base reactions which occur only under the strong acid conditions. The result will be a greater acidity and a lower observed field pH than is expected based on lab analysis.

The report goes on to discuss a concept of net acidity based upon the oxidation of sulphide by hydrogen peroxide, but as this concept has not been advanced beyond the most preliminary stage of investigation, I will not present it here.

The above analysis suggested potential for problems in current testing and assessment procedures. Further field investigations by Environment Canada in 1986 of quarries in the Lunenburg/Queens Co. of N.S. indicated pHs in slate quarries as follows:

pH	Standing Water	Running Water
6.5-7.5	0	0
5.5-6.5	5	5
4.5-5.5	3	3
3.5-4.5	12	7
2.5-3.5	21	8
	41 sites (90 samples)	23 sites (45 samples)

Thus it can be seen that current evaluation and approval procedures and testing methods for quarries and other developments on slate should be re-evaluated. A federal-provincial working group has been established and is addressing these issues on a province wide basis.

In support of the above need, two additional studies are currently being conducted for Environment Canada, one to establish techniques to identify high and low environmental risk zones within slate formations (Porter Dillon) and another to investigate in detail the hydrological character of surface and groundwater at two quarries adjacent the Hwy 102 interchange for Halifax International Airport (Technical University of Nova Scotia).

The overall emphasis of this paper is to reinforce the concept that prediction methods and their interpretation must be field applicable. Failure to field test and to know the limitations of such predictive tools can have very damaging results on the environment. Also, as the Halifax International Airport problem demonstrates, after-the-fact clean-up can be very expensive.

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Acid-Base Accounting
United States Coal Mine Spoils
and
Prep Plant Refuse

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Acid-Base Accounting is defined as a group of chemical and physical methods which characterize soil, rocks and mineral-producing wastes. These methods, which include sulfur content, neutralization potential, color, fizz and pH, are discussed individually and collectively with interpretation of acid-base data. Acid-toxic, potentially toxic and favorable overburden materials are defined in terms of components of the acid-base account. Emphasis is given to the selection of overburden materials which are ultimately used for reclamation purposes.

INTRODUCTION

The United States "Surface Mining Control and Reclamation Act of 1977" (Public Law 95-87) requires mining operations, prior to receiving a permit, to make provisions for the protection of the hydrologic balance both on and off the minesite. The criteria for making such provisions are usually dictated by state surface mining acts and regulations promulgated pursuant to such acts. At a minimum, regulations require a determination of the probable hydrologic consequences (PHC) of the proposed mining and reclamation upon the quality and quantity of ground and surface water.

The PHC is based on surface and groundwater baseline data from a proposed minesite. Regulations also require a geologic description of the permit area down to and including the lowest coal seam to be mined. The geologic description should include the chemical analysis of each stratum within the overburden to identify the horizons which contain potential acid-forming or alkaline-producing material (West Virginia Department of Natural Resources, 1985).

Geochemical data generated for these purposes are generally referred to as overburden analysis. The primary overburden analysis method used extensively throughout the United State and elsewhere is Acid-Base Accounting, which is the recommended method for coal overburden evaluation by the Office of Surface Mining and by environmental regulatory agencies in most of the coal producing states (Smith et al, 1976; Sobek et al, 1978; Sencindiver, 1981; Tettenburn, 1980).

ACID-BASE ACCOUNTING

Acid-Base Accounting is defined in this paper as a group of chemical methods which characterize soils, rocks, coal refuse, and wastes from mineral production as to their potential to produce acid or alkaline waters and soils when exposed by mining or other earth disturbance activities.

Strictly speaking, Acid-Base Accounting consists of two measurements: (1) either total sulfur or pyritic sulfur (where a more detailed quantification is necessary); and (2) neutralization potential. The accounting balances maximum potential acidity (from immediately titratable sources plus sulfuric acid equivalent calculated from total or pyritic sulfur) against total neutralizers (from alkaline carbonates, exchangeable bases, weatherable silicates or other rock sources capable of neutralizing strong acids as measured by the neutralization potentials).

Total sulfur content accurately quantifies potential acidity of materials when all sulfur is present as a sulfide mineral. Sulfide minerals include pyrite, marcasite, chalcopyrite and pyrrhotite, to name a few. Pyrite is the most common and abundant of the sulfide minerals and appears in many isometric crystal forms including cubic and octahedron. Marcasite (white iron pyrites) has the same chemical composition as pyrite; however, it differs from pyrite in crystal form. Marcasite is orthorhambic and commonly occurs in tabular crystals flattened parallel to the basal plane. Groupings of the crystals are often called spear pyrites and cockscomb pyrites (Berry and Mason, 1959). For this paper, sulfide minerals will be referred to as pyritic minerals or pyritic sulfur which will include marcasite, pyrite, and other species that can contribute to sulfur acidity.

When part of the sulfur occurs in nonacid-producing forms (sulfate and organic), the maximum potential acidity, as calculated, will be high. It is for this reason that calculations based on total sulfur are referred to as maximums, and that in doubtful cases, appropriate acid and water leachings should be made to rule out those forms of sulfur which do not produce additional acid. From the stoichiometric equation of pyritic sulfur oxidation, the maximum potential acidity can be calculated in terms of calcium carbonate equivalent (1% sulfur = 31.25 tons of calcium carbonate equivalent per 1,000 tons of materials).

The neutralization potential, the second component of a net acid-base account, measures the amount of neutralizers present in the overburden materials. This measurements is found by treating a sample with a known volume of standardized hydrochloric acid, heating to assure complete reaction, and titrating with a standard base. The result is then expressed in calcium carbonate equivalents, which can be negative if the sample is already acidic. As with total sulfur, the neutralization potential is a maximum value. When balanced against acidity from the total sulfur measurement, a net acid-base account can be stated.

SAMPLING PROCEDURES

Perhaps the most important step in the collection of geologic data is sampling. Overall, sampling includes the selection of overburden columns for physical and chemical analysis to aid in pre-mine planning, and the selection of samples from overburden columns to represent the individual rock units to be disturbed by mining.

Selection of Overburden Columns

When selecting overburden columns, it is important to identify and represent all geologic features that may affect the mining and reclamation of a site. These features include the properties of individual strata from the ground to below the lowest coal to be mined, fractures, faults, groundwater levels and degree of weathering (Smith and Freeman, 1978).

The number of overburden columns needed to represent a minesite depends upon geologic features, total area to be disturbed, mining methods, future land use, and the sensitivity of the area. This number can vary from a few in areas where correlation with existing data is possible to many, where large tracts of previously unmined areas are involved.

A general estimate, based on experience, suggests that analyzed columns should be spaced at intervals of one kilometer (0.6 miles), or less, in critical regions (Smith et al, 1976). However, the actual number should be selected on a site specific basis, based on topographic and geologic features. A large area mine with consistent stratigraphic features may require one to three columns for each 1,000 acres, where a mountain top or contour mine drained by a sensitive stream may require three or more columns in 100 acres. The important thing is to have sufficient columns to adequately characterize the overburden materials that are to be disturbed by mining and their influence on water quality and reclamation.

Sampling Geologic Columns

Before useful laboratory analysis can be performed, consistent sampling procedures must be utilized. Exploration cores can provide excellent samples. Since rock core remain intact, rock type can be accurately determined, depth from the surface and thicknesses can be measured, and vertical variation within a rock unit can be seen, noted, and sampled separately if the variation appears to be important.

The sampling of individual rock units to be disturbed by mining for chemical and physical analysis is ultimately involved in decisions about mining methods to insure efficient mining and environmental protection. Each sample selected must be identified as to its relative position in the minesite, and it must be representative of the sampling interval from which it was extracted.

If exploration cores are not available, satisfactory samples can be obtained by an experienced person using an air-rotary drill. Precise rock types, depths from the surface and some vertical variations within rock units may be lost; however, by using small sample intervals (1 to 3 feet depending on pre-drilling knowledge of the section) sufficient detail can be obtained. The chip samples can be composited to represent greater intervals prior to analysis if they appear to be chemically and physically uniform.

Selected (hand) samples can be taken from freshly exposed highwalls or other nearby excavations if neither core nor air-rotary drills are available. Accurate depths from the surface and variations in rock units can be determined satisfactorily; however, a fresh surface is necessary to minimize the effects of weathering on the analysis.

Materials of special interest can also be selectively sampled. Selective samples are especially useful for checking variations within rocks that appear similar and for determining properties of peculiar or extreme specimens.

Such samples include waste material from coal cleaning plants and composite samples for special analysis. When composite samples are prepared, care must be taken to represent each portion of the composite on a weight or volume basis.

LABORATORY METHODS

Successful laboratory methods provide data which allow important characteristics to be stated in terms of a few key parameters. In a broad sense, Acid-Base Accounting is a group of methods developed to characterize materials as to their potential to produce basic or acidic bodies and leachates when exposed by land disturbances. The acid-base account can be used on soil, rocks, coal refuse, wastes from mineral production, and other materials associated with land disturbances and reclamation. The parameters commonly measured are thickness, sample location, rock type, total sulfur, neutralization potential, color, fizz and paste pH (Sobek et al, 1978). When more detailed information is needed, sulfur fractionations are included. All results must be reproducible and adapted to correlation with field observations and measurements. The individual methods which comprise Acid-Base Accounting were developed cooperatively by geologists, soil chemists, and coal specialists to help improve coal mining efficiency and to create better land and water after mining.

As in all laboratory methods, attention must be given to all procedural steps to assure accurate and reproducible results within and between laboratories. Procedural steps include precise volume and weight measurements, allowing for complete chemical reactions, and reaching reproducible end points. Attention to procedural details, along with a quality control program, should assure reliable data.

According to established scientific sources (EPA 1979, Black 1965), there are three factors essential to a good quality control program: 1. Use only methods that have been studied collaboratively and found acceptable. These are usually published methods or variations of published methods as prescribed by federal and state agencies. 2. Analyze duplicate samples at regular intervals with each set of unknown samples to check analytical precision. To check accuracy, analyze standards with each set of unknown samples. Standards that closely approximate unknowns can be obtained from the National Bureau of Standards (NBS), the Environmental Protection Agency (EPA), other state and federal agencies, or prepared internally. 3. Analyze references samples several times a year to confirm the ability to produce acceptable results. Reference samples should be run whenever there is a change in procedures.

Color

Soil and rock colors shown in an acid-base account are determined by comparing air-dried, powdered (less than 60-mesh) samples with Munsell Soil Color Chart standards. These standards subdivide color into hue, value, and chroma. Hue is the dominant spectral color (red, yellow, green, blue and purple) and is related to wavelength of light; value is the measure of lightness or darkness and is related to total reflected light; and chroma indicates purity or strength of color (or departures from neutral of the same lightness). Color variables are expressed in order: hue, value/chroma (e.g. 10YR 6/3; hue = 10YR, value = 6, and chroma = 3).

Color variables can be used qualitatively in the laboratory or the field to estimate or predict rock and soil characteristics. Value can be used to distinguish highly carbonaceous black shales from true gray shales or other rocks that appear black. Bonecoal, roof shale, and carbonaceous rocks frequently contain significant amounts of pyritic materials and may be a source of high concentrations of sulfuric acid acidity unless neutralizers are present. Chroma can be used to distinguish intensely oxidized zones (brown, yellow, or red) from relatively unweathered (gray) zones that are usually deeper in the stratigraphic section. The oxidized zones are generally devoid of pyritic sulfur and high potential acidities, although they may have a wide range of pH values and of immediately titratable acid.

Fizz

Fizz is a visual indication of the presence of acid soluble carbonates in rock or soil samples. In the recommended standardized test, dilute hydrochloric acid (1:3) solution is dropped onto a powdered sample. The acid reacts with carbonates and a noticeable effervescence (bubbling) and sometimes even an audible "fizz" occurs. The bubbling and fizz is the release of carbon dioxide gas which is a product of the reaction.

The intensity of the fizz is related to the concentration and solubility of the carbonate in dilute acid at room temperature. Calcium carbonate (calcite or aragonite) is more soluble than calcium - magnesium carbonates (dolomite) or than mixed carbonates such as ankerite, and will react faster in dilute acid. The dolomitic reaction to acid can be intensified by heat or by using a stronger acid. It should be noted that highly soluble sodium and potassium carbonates can occur in rocks below the water table. However, these alkali carbonates do not seem to be common in American coal measures.

The fizz test was developed as a field clue and for laboratory use with the neutralization potential (N.P.) test described later. Fizz is a qualitative test and should be used and interpreted accordingly; but it can help to prevent gross errors in the N.P. determination.

Percent Total Sulfur

Total sulfur is measured to predict the amount of bases that would be required to neutralize the potential acid possible from the oxidation of pyritic sulfur. The sample is heated to high temperatures in a stream of oxygen and, as sulfur dioxide is released, it is measured amperometrically. From the stoichiometric equation of pyrite oxidation, the amount of base needed to neutralize the resultant acid can be calculated in tonnes of calcium carbonate equivalent (1% sulfur = 31.25 tons $CaCO_3$ equivalent per 1,000 tonnes of material).

Neutralization Potential (N.P.)

The concentrations of neutralizing bases present in a soil or rock sample are found by treating the sample with a known excess of standardized hydrochloric acid (Jackson, 1958). Unreacted acid is then titrated with standardized sodium hydroxide. The amount of acid consumed in the reaction is measured and converted to neutralization potential. The neutralization potential is expressed as calcium carbonate equivalent because calcium carbonate is the most abundant neutralizer in sediments associated with coal. Heating the sample-acid mixture to insure complete reaction and back titrating to a stable pH of 7.0 (end point) gives maximum neutralization potential.

Paste pH

A paste or slurry is prepared by mixing powdered soil or rock in distilled water. The soil/water ratio is approximately 2:1 and is preferred over traditional 1:1 ratio because it assures close contact between the soil and the electrode. Materials with paste pH values of less than 4.0 are defined as acid toxic. Values below 4.0 indicate immediately titratable acidity and possibly toxic levels of aluminum and other acid-soluble elements. The immediate acidity must be considered when calculating neutralization requirements. Negative neutralization potentials are a good estimate of immediate lime requirements (Sturey, 1979).

Sulfur Forms

Total sulfur content accurately quantifies potential acidity of materials when all sulfur is present as a pyritic mineral. When gypsum is present in the rock column, or the materials are weathering from recent exposure, sulfur occurs in the form of sulfates. Samples high in organic carbon usually contain organic sulfur. Sulfate does not produce additional acid by oxidation and organic sulfur does not oxidize fast enough to cause significant sulfate acidity. Appropriate acid and water leachings can be made to determine these sulfur forms.

Two samples are reacted with acid and then washed with water prior to sulfur determination as described under Percent Total Sulfur (above). One sample is reacted with hydrochloric acid and then washed with water. The HCl-extractable (sulfate) sulfur is calculated by subtracting the sulfur remaining in the sample from the original total sulfur. The second sulfur sample is reacted with nitric acid and washed with water. The sulfur remaining in this sample is the nonextractable (organic) sulfur. Pyritic sulfur is calculated by subtracting the HCl extractable and the nonextractable sulfur from the original total sulfur.

INTERPRETATION OF DATA

The problem of acid-mine drainage and correlated conditions can be vastly simplified and brought under technical control by realizing that the entire complex of variables and what can be done about them depends upon how much acid-forming material and how much neutralizing material we have to deal with. Acid-Base Accounting quantifies acid-forming and neutralizing materials by relatively simple laboratory measurements. The results can be reasonably identified with recognizable rocks and earth materials on the minesite. Thus, we can pin down the location and ultimate magnitude of acid sources on the one hand, and of neutralizing materials on the other. With only very rare exceptions can we be confident that toxic elements, notably iron and aluminum in solution, constitute a serious problem in conjunction with strong acids; whereas, these metals are essentially insoluble and harmless near neutrality or in calcium-controlled alkalinity.

Acid-Base Accounting not only indicates where and how much acid and neutralizers are available for use, but it identifies where the materials occur in the coal or overburden material. It also provides a basis for judging how to minimize the acid by controlled excavation and placement, and how to make the most of the neutralizers.

The practice of detailed sampling and analysis of coal overburdens to provide for prevention of acid-drainage is a relatively new and specialized field which is not covered intensively in most academic circles, even where theories of sedimentary environments have been emphasized. We must realize that some aspects of sedimentary environments remain controversial, and many earth science phenomena have influenced coal swamps since initial sedimentation took place. Diagenetic, tectonic, leaching, ancient and recent weathering and completely unknown events may account for the present detailed properties of coal overburdens that are involved in determining acid-mine drainage potentials. It cannot be assumed that present properties are all determined by the conditions prevailing during initial sedimentation, nor that we know all the pathways by which our near-surface rocks and earthy materials have reached their present state in our environment. We must deal with what we find in the coal fields, and apply the best scientific knowledge that we have at hand.

Acid-Toxic and Potentially Toxic Material

From Acid-Base Accounting, potentially toxic material is defined as any rock or earth material having a net potential deficiency of 5.0 tonnes of calcium carbonate equivalent or more per 1,000 tonnes of material. The 1,000 tonnes are based on the practical assumption that an acre plow-layer contains 1,000 tonnes of soil. Regardless of the acid-base account, materials which have a pH value of less than 4.0 in a pulverized rock slurry in distilled water are defined as being acid-toxic.

The choice of the deficiency of 5 tons of calcium carbonate equivalent per 1,000 tonnes of material as the division between toxic and non-toxic material is somewhat arbitrary and is based on a pH of neutrality. If rock or soil samples were defined to be toxic at much lower calcium carbonate equivalent deficiencies than 5 tonnes per 1,000 tonnes, many of our native soils would be considered toxic. On the other hand, with deficiencies much greater than 5 tonnes per 1,000 tonnes, toxic concentrations of plant-available aluminum with pH values below 4.0 often develop rapidly.

Favorable Material

Favorable material is defines as rock or earth material that will support desirable vegetation and assure satisfactory water quality and land resources both during mining and post-mining. Materials used on the surface for plant establishment and growth should have paste pH values between 5.5 and 7.5 as well as medium to high levels of plant nutrients. They should also weather to form sufficient fines to produce a soil with medium texture, permeability, and favorable water-holding capacity.

When acid-toxic or potentially toxic materials are to be returned to the backfill, neutralizing material is needed to prevent acid-production from sulfide mineral oxidation. Materials to be mixed with acid toxic, or potentially toxic materials should have excess $CaCO_3$ equivalent and should break-down rapidly to provide maximum surface area when exposed to acid. These mixtures should then be surrounded, sandwiched, or otherwise sealed with material that compacts readily to impede water and air from reaching the acid-producing material.

Acid-Base Account and Bar Graph

The results of Acid-Base Accounting of a geologic column are shown in tabular form in Table 1 and graphic form in Figure 1 and are referred to as an Acid-Base Account and an Acid-Base Graph, respectively. A brief summary of the data follows.

There are four potentially toxic horizons (Samples 4, 5, 9, and 18) and one acid-toxic zone (Sample 12) in this overburden column as shown by the "Max. Needed" values greater than 5.0 tonnes per 1,000 tonnes of $CaCO_3$ equivalent deficiency. They also appear on the Acid-Base Graph as bars to the left of the 5.0 tonne deficiency line (dashed vertical line). These are the zones of major concern during mining and reclamation. These strata are dark with low color values (mostly 4 or less) and should be easily recognized during mining.

Tables 2 and 3 give Sulfur Forms and an Acid-Base Account calculated with pyritic sulfur for acid-toxic and potentially toxic strata. Sample 12 remains acid-toxic as

							Calcium Carbonate Equivalent Tonnes/1000 Tonnes				
Sample Number	Depth (Meters)	Strata Thick. (m)	Rock Type	Fiz	Color	% S	Max. From % S	N.P. $CaCO_3$ Equiv.	Max. Needed (pH-7)	Excess $CaCO_3$	Paste pH
1	1.22	1.22	Soil	0	10yr 7/4	.048	1.50	1.03	.47		4.7
2	2.44	1.22	MS	1	10yr 7/4	.058	1.81	10.82		9.01	6.2
3	4.27	1.83	MS	2	10yr 7/2	.145	4.53	31.30		26.77	7.4
4	5.49	1.22	MS	1	10yr 5/2	.789	24.66	5.16	19.50		5.2
5	6.71	1.22	MS	0	10yr 4/2	.508	15.88	10.67	5.21		6.3
6	7.92	1.22	MS	0	10yr 5/1	.649	20.28	18.60	1.68		6.5
7	10.97	30.48	SiS	1	5y 6/1	.411	12.84	18.21		5.73	7.3
8	12.80	1.83	SS	3	10yr 6/1	.311	9.72	56.14		46.42	7.8
9	13.41	.61	MS	0	10yr 4/1	.679	21.22	12.28	8.94		6.6
10	15.85	2.44	MS	1	10yr 5/2	.536	16.75	19.47		2.72	7.5
11	18.29	2.44	SiS	0	10yr 5/2	.385	12.03	15.07		3.04	7.2
12	18.90	.61	Coal-MS	0	10yr 3/1	.808	25.25	-1.05	26.30		3.7
13	19.51	.61	MS	0	10yr 4/1	.307	9.59	12.43		2.84	5.7
14	21.34	1.83	MS	1	10yr 5/1	.096	3.00	17.09		14.09	7.7
15	22.55	1.22	SS	1	10yr 6/1	.113	3.53	15.00		11.47	7.9
16	24.99	2.44	SS	1	10yr 6/1	.056	1.75	16.86		15.11	7.4
17	25.60	.61	MS	0	10yr 5/1	.327	10.22	11.87		1.65	6.9
18	26.82	1.22	MS	0	10yr 4/1	1.03	32.19	11.33	20.86		6.0
19	28.04	1.22	MS	0	10yr 4/1	.256	8.00	14.85		6.85	7.6
20	29.26	1.22	MS	0	10yr 3/2	.476	14.88	13.13	1.75		7.1

Table 1 ACID-BASE ACCOUNT

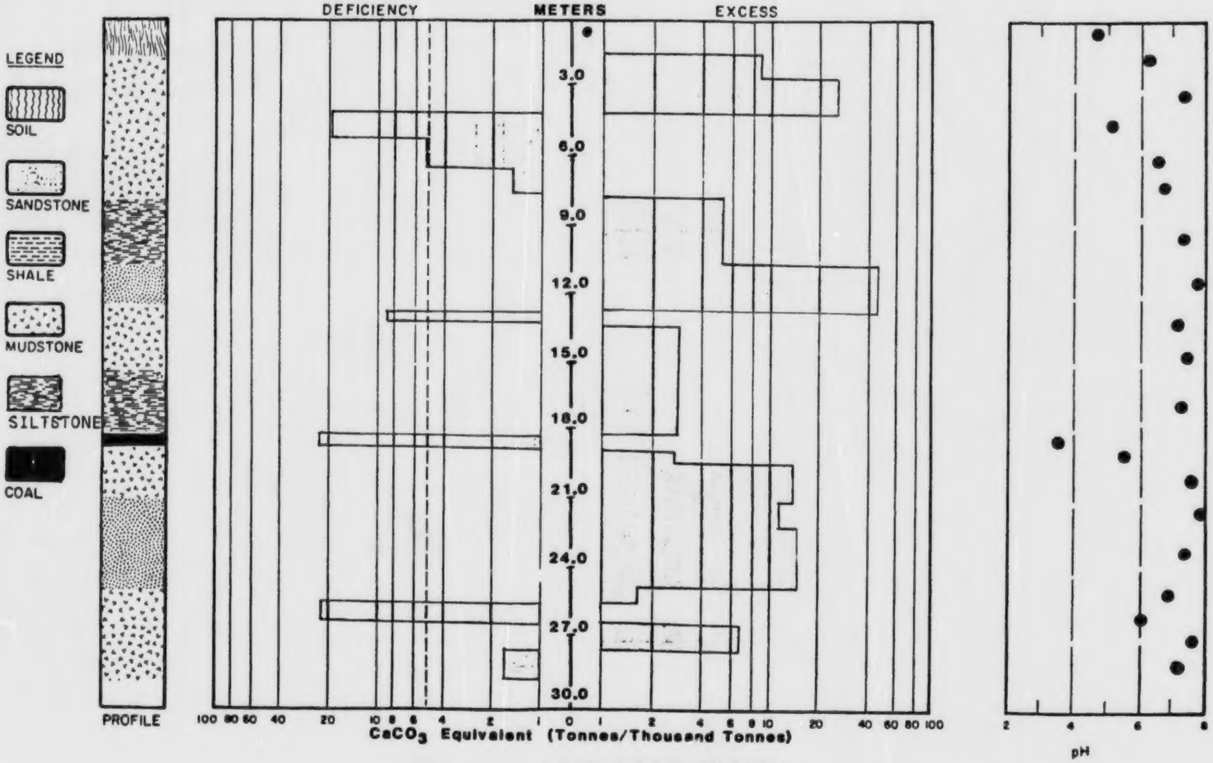

Figure 1 ACID-BASE GRAPH

defined by pH less than 4.0. The low pH, high sulfate and low pyritic sulfur content indicate extensive oxidation prior to chemical analysis. Only two of the four potentially toxic strata (Samples 4 and 18) remain potentially toxic based on pyritic sulfur.

Sample No.	Total %	Pyritic %	Sulfate %	Organic %
4	.789	.636	.102	.051
5	.508	.459	.049	<.005
9	.679	.483	.063	.133
12	.808	.017	.298	.493
18	1.03	.690	.191	.149

TABLE 2 - SULFUR FORMS

The favorable materials in this overburden have relatively low sulfur values, high pH values, and excess $CaCO_3$ equivalent. These rocks can be used to mix with and/or to surround potentially toxic material to be returned to the backfill after mining. The fizz values in the acid-base account indicate the carbonates in these rocks are mostly calcium carbonate and should be available for rapid neutralization of sulfuric acid. The rate of solubility of carbonates depends upon particle size, reactive surface area, temperature, and degree of acidity of the water. Reaction rates increase substantially in carbonic acid and

							Calcium Carbonate Equivalent Tonnes/1000 Tonnes				
Sample Number	Depth (Meters)	Strata Thick. (m)	Rock Type	Fiz	Color	Pyritic % S	Max. From % S	N.P. $CaCO_3$ Equiv.	Max. Needed (pH-7)	Excess $CaCO_3$	Paste pH
4	4.27-5.49	1.22	MS	0	10yr 5/2	.636	19.88	5.16	14.72		5.2
5	5.49-6.71	1.22	MS	0	10yr 4/2	.459	14.34	10.67	3.67		6.3
9	12.80-13.41	.61	MS	0		.483	15.09	12.28	2.81		6.6
12	18.29-18.90	.61	Coal-MS	0	10yr 3/1	.017	.53	-1.05	1.58		3.7
18	25.60-26.82	1.22	MS	0	10yr 4/1	.690	21.56	11.33	10.23		6.0

Table 3 ACID-BASE ACCOUNT

is very high in mineral acids. Maximum surface area of calcareous material can be generated by blasting and/or material handling to expose inherent carbonates.

SUMMARY

After proper sampling, sample preparation and chemical analysis, overburden materials can be characterized by Acid-Base Accounting. In all steps leading to an acid-base account, adequate quality control is essential to reliable interpretation of results. Interpretation of an acid-base account requires an inter-disciplinary viewpoint, including the disciplines of chemistry, ecology, geology, and plant and soil science.

From the definitions of acid-toxic, potentially toxic and favorable material, acid-producing and neutralizing materials can be located in an overburden by Acid-Base Accounting. Knowing where these materials occur can aid decisions on mining and reclamation and prevent acid-water and vegetation problems.

After Acid-Base Accounting and interpretation by an experienced person, there may still be doubt about the potential for acid-production from specific strata or coal wastes. Doubtful cases include materials with high potential acidity from pyritic sulfur and only enough neutralizer to react with the acidity. Simulated weathering (leaching) studies may give useful information for prediction of acid generation. Such studies should be based on backfill and material handling scenarios established from pre-mine planning. Materials to be placed in the backfill or subject to special handling and placement should be leached in proper proportions to simulate position in the backfill or permanent disposal areas. The exact experimental design will depend upon the backfill or material handling procedures that are planned for a specific site. Designs of the leaching columns should contain similar proportions and arrangements of the materials as the proposed backfill (Sturey et al, 1982; Hanna and Brant, 1962).

Data which are generated from leaching columns are extremely difficult to equate to real life applications. In all studies, Acid-Base Accounting characterization should be the foundation by which preliminary observations and evaluations are formulated. While attempts have been made to alter some of the individual steps comprising Acid-Base Accounting, to date, there are no alternatives that sufficiently evaluate overburden characteristics (Shelton, et al, 1984).

Prediction of Acid Mine Drainage from Duluth Complex Mining Wastes in Northeastern Minnesota, U.S.A.

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Abstract

Ten Duluth Complex drill core samples, with sulfur contents ranging from 0.47 to 2.17 percent, were leached in a laboratory experiment. The variation in sulfur content was due largely to fluctuations in iron sulfide content. Solids containing 0.8 percent sulfur or less produced neutral drainage while, with one exception, solids with sulfur contents of 0.92 percent or more produced acidic drainage. The results indicated that acid production due to oxidation of sulfide minerals containing iron, such as pyrrhotite, increased linearly as the sulfur content increased. The acid consumption due to silicate mineral dissolution was relatively constant despite minor variations in the silicate mineralogy. A sample containing 1.17 percent sulfur produced neutral drainage due to the buffering provided by a calcium carbonate content of three percent. Leaching of a sample from a test stockpile indicated that the fine fraction of stockpile solids exerted a major influence on the drainage chemistry. Laboratory results from a solid containing 0.92 percent sulfur bore a resemblance to field data from a test stockpile with a bulk sulfur content of about 0.6 percent, although additional data are required for accurate extrapolation of laboratory results to a field setting.

Acknowledgements

The laboratory experiments were conducted by David Antonson, whose diligence and perceptiveness improved the experimental efficiency and accuracy. Albert Klaysmat, with assistance from Jean Drotts, was responsible for the prompt and precise chemical analysis of the solids and numerous drainage samples generated. Paul Pojar and Milan Dicklich contributed to the construction of the experimental apparatus. Permission for use of the drill core samples was granted by Kennecott Minerals Company.

Introduction

The Duluth Complex, located in northeastern Minnesota, is an intrusive formation containing low grade copper and nickel sulfides (Listerud and Meineke, 1977). These deposits comprise 25 percent of the United States' copper resource and the nation's largest nickel sulfide resource (Minnesota Environmental Quality Board, 1980; Kingston et al., 1970). Development of these resources would require open pits which will eventually fill and discharge water, possibly containing elevated concentrations of trace metals, to the environment (Thingvold et al., 1979).

Based on conceptual models, the open pits could produce an estimated 4 to 10 square kilometers of waste rock stockpiles (Sturgess, 1981; Veith, 1978), which would generate an estimated runoff of 1.5 to 3.8 billion liters in a year of average precipitation (Hewett, 1980). Trace metal concentrations in runoff from waste rock stockpiles at the Erie Mining Company Dunka Mine were 10 to 10,000 times natural background levels of streams in the area (Eger et al., 1981). Since these metals have been shown to be toxic at low aqueous concentrations (United States Environmental Protection Agency, 1976), means of mitigating mining drainage impacts will be required for resource development.

The degree of mitigation required may be a function of the potential for trace metal and acid release from a given stockpile. This potential is dependent upon, among other factors, the composition of the stockpiled rock. Examining dissolution with respect to all chemical and mineralogical variables would, at best, be extremely complicated. Characterizing mine wastes with respect to these variables in the course of operations is also impractical. Consequently a more simplistic course of experimentation, using sulfur content as the independent variable, was adopted.

The effect of sulfur content on the leaching of Duluth Complex mining wastes has been examined for a limited number of samples. Field tests indicated that over a six year period, test stockpiles containing 0.8 and 1.4 percent sulfur produced acid leachate while piles containing 0.6 percent sulfur did not (Eger and Lapakko, 1985). Subsequent data indicate the drainage pH has at times decreased below pH 5 at one of the four piles containing 0.6 percent sulfur. In shake flask tests, neutral leachate was generated by tailings containing 0.92 percent sulfur or less, while acidic leachate was generated by tailings with sulfur contents equaling or exceeding 2.65 percent (Natarajan and Iwasaki, 1983).

The rate of metal sulfide oxidation has been reported to be directly proportional to the reactive sulfide mineral surface area available (Lapakko, 1980; Nelson, 1978; Sato, 1960a, 1960b; Sato and Mooney, 1960). In turn, the available surface area of nonporous solids is a function of the sulfide content of the rock, particle size, and the roughness of the surface. Since the copper and nickel content of mine wastes is small, the sulfur content is an indicator of iron sulfide minerals, which are of major importance due to their role in acid production.

Since other sulfides are present in limited quantities, their influence is diminished and to some extent uniform. By using uniformly small solids, the influence of particle size and host rock association of the iron sulfides is normalized. Since the sulfides are liberated in small size fractions, the sulfur content will represent the available sulfide surface area, assuming specific surface area does not vary greatly among sulfide minerals of the same particle size. Since all the sulfide surface area is known to be available, the data can be extrapolated to mining wastes based on their available sulfide surface area. This extrapolation will allow appropriate mitigation based on the rock composition.

The use of sulfur content as the independent variable neglects the effects of many other variables, environmental as well as compositional, on the leaching of mine wastes. Although there is a given degree of compositional uniformity within an ore body, there are deviations from the norm. To accurately assess the leaching potential of mining wastes from a given body, a large number of samples must be analyzed. This implies the use of efficient and accurate laboratory techniques from which results can be extrapolated to operational conditions. This study involved examination of the leaching behavior of a small number of Duluth Complex samples with variable sulfur contents, and determination of the feasibility of extrapolating laboratory results to operational conditions.

Methods

Materials

An experiment was conducted to examine the effect of solid phase sulfur content on the quality of drainage from Duluth Complex mining wastes. The sulfur content of the 11 samples examined ranged from 0.47 to 2.17 percent. Acid digestion (U.S. Bureau of Mines, 1980) and subsequent analysis revealed typical solid phase copper and nickel contents of 0.13 to 0.23 and 0.05 to 0.09 percent, respectively (table 1). The samples were selected to approximate mining wastes, based on a 0.2 percent copper cutoff. Plagioclase, olivine, pyroxenes, and biotite were the predominant minerals (table 2).

With one exception, the solids were MINNAMAX core samples which had been drilled from 1974 to 1977. The core samples

Table 1. Elemental composition of solids, values in weight percent.

S	Cu	Ni	Co	Zn	S with FeS^1	Pct of total S bound by Fe
0.47	0.23	0.060	0.011	0.017	0.31	65
.59	.13	.072	.013	.0070	.44	75
.80	.19	.067	.011	.010	.66	82
.92	.091	.049	.014	.016	.83	90
1.17	.23	.082	.011	.015	1.0	85
1.24	.19	.066	.012	.013	1.1	91
1.35	.23	.086	.018	.011	1.2	87
1.87	.17	.088	.021	.011	1.7	92
2.01	.20	.055	.012	.020	1.9	93
2.17	.20	.078	.010	.027	2.0	93
1.26[2]	.55	.20	.018	.014	.97	77

[1] Calculated as the sulfur not bound by Cu, Ni, Co, or Zn assuming all trace metals were bound by sulfur in a 1:1 mole ratio of sulfur to metal.

[2] Sample from test pile.

had been crushed to minus 1/4 inch, placed into plastic bags, and stored inside 55 gallon barrels at the MINNAMAX site near Babbitt, Minnesota. Under the direction of Karl Smith, the samples were processed for experimental use at the Mineral Resource Research Center at the University of Minnesota. The plus 10 mesh fraction was selected for experimentation in order to minimize the amount of surface area exposed to oxidation while the samples had been stored after drilling. This fraction was stage crushed using a pulverizer and wet-screened to segregate the -100/+270 mesh fraction for use in laboratory tests.

Table 2. Approximate modal compositions of Duluth Complex Samples. (By Tanya Sabelin, Mineral Resources Research Center, University of Minnesota).

Mineralogy (%)

PCT S	Olivine	Pyroxene	Plagioclase	Biotite	Opaque	Other	Calcite
0.47	30	10	45	5	8	2	-
.59	17	4	68	2	5	4	-
.80	20	8	59	4	7	2	-
.92	31	27	24	2	16	-	-
1.17	19	15	28	2	8	25	3
1.24	17	27	47	1	5	3	-
1.35	10	14	57	-	9	10	-
1.87	18	4	57	2	10	9	-
2.01	7	15	59	6	13	-	-
2.17	15	8	51[1]	6	17	3	-
1.26[2]	15	11	49	3	16	4	-

[1] Includes plagioclase and quartz; of this total maybe as much as 1/3 to 1/2 is quartz.

[2] Sample from test pile.

The remaining sample (1.26 pct S) was collected from a test stockpile which was constructed in April 1977 at the MINNAMAX site. The pile was constructed of rock mined during the development of a test shaft. When part of the test pile was removed in September 1982 a representative sample, weighing over three tons, was collected for analysis of particle size distribution, specific surface area, chemistry, and mineralogy. A subsample in the -100/+270 mesh fraction was collected for use in the experiments.

Procedure

Several techniques have been used in laboratory leaching of mining wastes (Caruccio, 1986; Ferguson and Mehling, 1986).

Figure 1. Project site in northeastern Minnesota.

The method used was based on the principle that sulfide minerals oxidize in the presence of atmospheric oxygen and water (Gottschalk and Buehler, 1912; Caruccio et al., 1980). The experiments were conducted at the site previously leased by MINNAMAX, which is presently leased by Kennecott Minerals (figure 1). Samples of -100/+270 mesh rock were rinsed with 200-ml volumes of distilled deionized water to examine the water quality of the effluent. The 200 ml of rinse water was allowed to remain in contact with the solids for five minutes and filtered.

At the beginning of the experiment, the solids were rinsed five times to remove oxidation products generated after crushing and wet sieving. After this initial washing the solids were rinsed weekly. Four rinses were used after one week, three rinses after two weeks, and two rinses were used in subsequent weeks. The rinse water, or drainage, was analyzed for specific conductance, pH, sulfate, copper, nickel, cobalt, zinc, calcium, magnesium, sodium and potassium. Specific conductance was analyzed using a Myron L conductivity meter, while a Radiometer 29 pH meter was used for pH analyses. Sulfate was analyzed using either the barium sulfate turbidimetric technique (AHPA, 1975) or a Technicon autoanalyzer. Metals were analyzed with a Perkin Elmer 603.

Ten of the solids were run in duplicate while single reactors were used for the remaining four solids (0.47, 0.59, 1.35, and 2.01 pct S). Between rinses the solids were stored in a box to dry. A cover was placed about three cm above the upper edge of the box to allow drying of the solids and prohibit the input of airborn debris. A thermostatically controlled heating pad was placed beneath the box to maintain a constant temperature. Water containers were placed in the box in an attempt to maintain a fairly constant humidity. Temperature and relative humidity in the box were monitored four to five times per week. The average temperature was 25°C with a standard deviation of 1.0°C, while the corresponding values for relative humidity were 56 and 9.1 percent.

Results and Discussion

Laboratory Data

Drainage pH generally decreased as the solid phase sulfur content increased and as the duration of leaching increased.

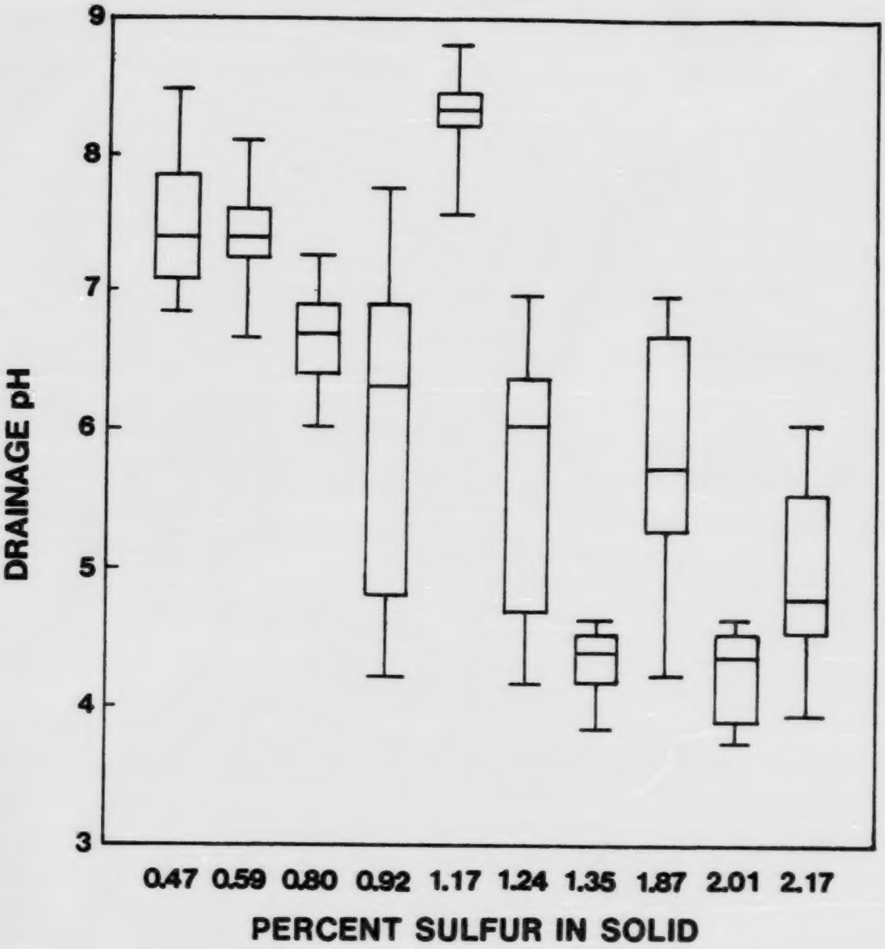

Figure 2. Drainage pH versus sulfur content of the solid phase. Horizonal lines 25, 50 and 75 percentiles, as well as the minimum and maximum values during the experiment.

The pH of drainage from solids containing less than 0.9 percent S was consistently above 6, while drainage from solids containing more than 1.3 percent S was typically below pH 6. Between 0.9 and 1.3 percent S was a transition zone. Drainage pH from one of the three transitional solids continuously remained above 7.5 (figure 2). The drainage pH from the other two transitional solids was initially in the neutral range, but steadily decreased to about 4.2 where it plateaued. The tendency for drainage pH to decrease and level over time was observed with most solids, but the decrease was most pronounced with these transitional solids. These pH trends were the net result of acid production by the oxidation of iron sulfide minerals and acid consumption by the dissolution of silicate minerals.

The dissolution of iron sulfide minerals leads to the production of acid, as is indicated by reaction 1 (Nelson, 1978). The iron sulfide oxidation may be influenced by

$$\mathrm{FeS(s)} + (3/2)\mathrm{H_2O} + (9/4)\mathrm{O_2} = \mathrm{FeOOH(s)} + 2\mathrm{H^+(aq)} + \mathrm{SO_4{}^{2-}(aq)} \qquad (1)$$

microbial activity (Natarajan et al., 1982), galvanic interactions with other sulfide minerals (Natarajan and Iwasaki, 1983), as well as other factors. The oxidation of sulfide to the hexavalent sulfur present in sulfate does not contribute acid. The two moles of acid produced are the net result of the oxidation of ferrous iron and the subsequent precipitation of ferric iron, as lepidocrocite for example (Nelson, 1978; Sung and Morgan, 1980).

The rate of iron sulfide oxidation was reflected by the appearance of sulfate in solution. Under the conditions in this experiment, and in any oxidizing environment, sulfate is the ultimate reaction product of sulfide oxidation. Over the aqueous concentration range observed in the laboratory samples, sulfate will remain in solution as opposed to precipitating. Therefore, at steady state, the rate of sulfide oxidation equals the rate of appearance of sulfate in solution. Since 82 to 93 percent of the sulfide present in these samples was bound by iron (table 1), oxidation of these iron sulfides was most likely the major sulfate source. Pyrrhotite is the predominant iron sulfide in the Duluth Complex (Bonnichsen, 1972), and in these samples.

The rate of sulfate appearance was proportional to the mean sulfate concentration, since the rinse volumes were essentially equal, the mass of each of the solids leached was 75 grams, and the rinse interval was constant. Mean concentrations from weeks six through seventeen were used since a) sulfate concentrations during the initial five weeks were typically low and not representative of the sulfate release in general, and b) after the inital seventeen weeks, the rinsing interval for some of the solids was extended beyond the standard duration of one week, which affected the rate of sulfate release.

The rate of iron sulfide oxidation, as reflected by mean sulfate concentrations, correlated highly with the solid phase sulfur content; and the degree of correlation increased when minor variations in specific surface area among the samples were taken into account. The percent sulfur in the solid phase accounted for 77.6 percent of the variation in the rate of iron sulfide oxidation (r = 0.776). The r value increased to 0.963 when two outlying data points were ignored. The anomalous sulfate concentrations, from the solids containing 1.35 and 1.87 percent S, were higher than predicted by the linear model describing the eight remaining points (figure 3). This deviation was likely due to minor variations in specific surface area among the solids.

Although all solids were in the -100/+270 mesh size fraction, limiting particle diameter to the range of 0.053 to 0.149 mm, there were minor variations in specific surface area. The specific surface areas, as determined by nitrogen adsorption, of the two anomalous solids were 65 to 80 percent higher than the average of the remaining solids. To account for this variation, the available sulfide surface area per gram solid was estimated as the product of the sulfur content and the specific surface area. The correlation coefficient was 0.846 for the linear variation of sulfate release rate with respect to this available sulfide surface area. This dependence is consistent with a surface reaction dominating the sulfide oxidation.

The r value increased to 0.962 when the data for the solid containing 1.17 percent sulfur was omitted. Sulfate release

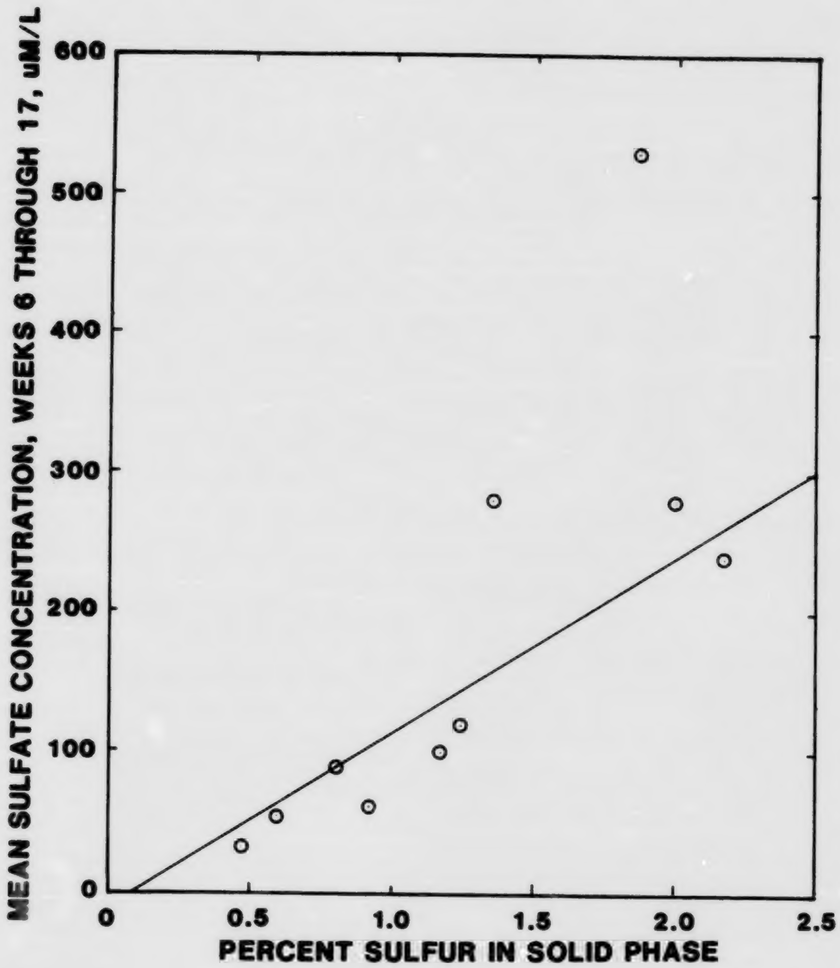

Figure 3. Sulfide oxidation rate, as represented by mean sulfate concentration, versus percent sulfur. Points at 1.35 and 1.87 percent S were excluded from regression analysis of line shown.

from this solid was lower than predicted by the linear model describing the remaining nine data points. This was the only sample containing calcium carbonate, which may have been solely responsible for the elevated surface area of this sample. The elevated pH of the drainage from this sample may also have inhibited the rate of sulfide oxidation (Nelson, 1978).

Some of the acid generated was consumed, or neutralized, by the dissolution of silicate minerals. Plagioclase, olivine, and pyroxenes are the silicate minerals which comprise the majority of the host rock. The dissolution of anorthite, which is a member of the plagioclase solution series, is represented by the following reaction (Stumm and Morgan, 1981).

$$\begin{aligned} &CaAl_2Si_2O_8(s) + 2H^+(aq) + H_2O \\ &\qquad = Ca^{2+}(aq) + Al_2Si_2O_5(OH)_4(s) \end{aligned} \tag{2}$$

The dissolution of these minerals in laboratory experiments has often been reported as parabolic with respect to time. The initial and more rapid release involves the reversible exchange of alkali ions on the mineral surface with hydrogen ions in solution (Holdren and Berner, 1979; Busenberg and Clemency, 1976). The rate of release subsequently decreases, eventually becoming linear with respect to time (White and Classen, 1979; Busenberg and Clemency, 1976). The rate of dissolution during this phase is relatively slow (Siegel, 1981; White and Classen, 1979; Busenberg and Clemency, 1976).

Dissolution of the silicate minerals was rapid enough to neutralize the acid generated by iron sulfide oxidation in the solids containing 0.8 percent S or less. The pH of drainage from these solids remained above 6. For the solids containing at least 1.35 percent S, reflecting the higher content of iron sulfides, the acid production rapidly overwhelmed the buffering capacity. The drainage from the solids containing 0.92 and 1.24 percent S was initially neutral but gradually decreased, indicating that the rate of iron sulfide oxidation ultimately exceeded the dissolution rate of the silicate minerals present.

The balance of acid production by iron sulfide oxidation and acid consumption by silicate mineral dissolution is illustrated semiquantitatively by comparing sulfate concentrations with the sum of calcium and magnesium values. The release of one mole of sulfate represents the production of two moles of acid, while the release of one mole of calcium or magnesium represents the consumption of two moles of acid (reactions 1 and 2, respectively). Initially the sum of calcium and magnesium concentrations exceeded the sulfate concentration, thus the ratio of the sum to sulfate was greater than one. This indicates that the rate of acid consumption by silicate mineral dissolution initially exceeded the rate of acid production by iron sulfide oxidation. This is born out by the fact that the drainage pH was initally in the neutral range (figure 4).

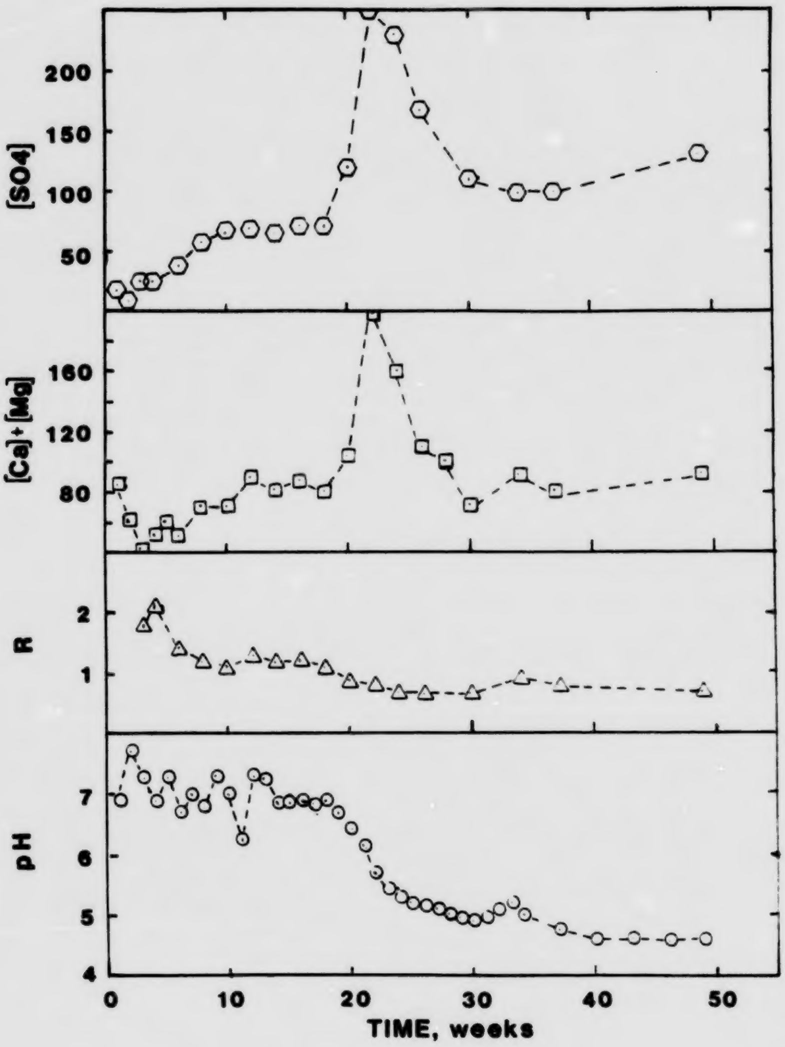

Figure 4. Drainage from 0.92 percent S solids versus time. Concentrations are in micromoles per liter, and R is the ratio of the sum of Ca and Mg concentrations to the sulfate concentration.

The rate of acid production gradually increased over the first 18 weeks, but the rate of acid consumption increased concomitantly and neutralized the acid produced. After 20 weeks the acid production increased dramatically and the silcate mineral buffering was overcome. At this time, the ratio of the sum of calcium and magnesium concentrations to sulfate concentration dropped below a value of one and pH decreased, ultimately reaching 4.5. The sum of the calcium and magnesium concentrations during this phase was lower than those in the initial phase, reflecting the decreasing rate of silicate mineral dissolution reported for laboratory experiments using isolated mineral phases (Siegel, 1981; White and Classen, 1979; Busenberg and Clemency, 1976).

The buffering supplied by the silicate minerals was fairly uniform among the solids, despite differences in silicate mineralogy (table 2). Apparently these differences had little effect on the overall rate of silicate mineral dissolution. The sample containing 1.87 percent S did exhibit slightly greater buffering and calcium release, and its composition is being examined in greater detail.

The uniformity in buffering among the silicate minerals is particularly important in predicting drainage pH. The primary copper-nickel mineralization of the Duluth Complex occurs as disseminated sulfides in a troctolitic-gabbroic series of rocks which intrudes an anorthositic and felsic series of older rocks (Weiblen and Morey, 1975). About 90 percent of the host rock is composed of silicate minerals, predominantly plagioclase, olivine, and pyroxenes (Stevenson et al., 1979). If the buffering supplied by these minerals is somewhat uniform, the drainage pH will be controlled by the rate of acid generation due to oxidation of iron sulfides. Pyrrhotite is the predominant form of these iron sulfides (Bonnichsen, 1972). Determination of the oxidation rate based on sulfur content will allow prediction of the acid producing potential of mining wastes based on their sulfur content. Investigation of a wider variety of rocks will lend further insight into the influence of the various silicate minerals on buffering, as well as other factors.

In contrast to the general uniformity in buffering, the solid containing 1.17 percent S, despite exceeding the sulfur content of one of the acid solids, produced drainage pH values which were typically above pH 8. Mineralogical analysis of this solid indicated a calcium carbonate content of about three percent. Dissolution of this calcium carbonate was reflected by calcium concentrations in drainage from this solid which were three to five times those in drainage from all but one other solid. Due to its rapid dissolution, the small amount of calcium carbonate was able to provide more buffering than the silicate minerals, which typically comprised more than 80 percent of the solids. This further emphasizes the slow kinetics of, and limited buffering by, dissolution of the silicate minerals.

The effectiveness of the silicate mineral buffering may have been further limited by the physical factors in the reaction environment. Iron sulfides can be oxidized by the oxygen and water vapor present in the air, while silicate dissolution due to the presence of oxygen and water vapor is negligible. The acid resulting from iron sulfide oxidation may contact only the silicate minerals in the immediate vicinity of the iron sulfides. Dissolution of these silicates can leave a less reactive solid phase such as kaolinite (reaction 2, and Busenberg, 1978), which may further inhibit the transport of acid to other mineral surfaces and further limit buffering. This inhibition of buffering reactions may contribute to the increased metal recovery associated with the "resting" of leach columns and dumps in metallurgical work.

In a submerged reaction environment, such as that in batch reactors, the acid generated could be readily transported to silicate mineral surfaces via the aqueous medium. As a result, batch reactor data may overestimate the buffering available in an environment subject to wet-dry cycling, such as stockpiles and unsubmerged mine walls.

As drainage pH decreased trace metal release increased, and this increase was quantified by linear regression analysis of the base 10 logarithm of metal concentration versus pH. For the log copper concentration versus pH plot, the slope was -1.11 (r=-0.944, n=38), and the corresponding slope for nickel was -0.99 (r=-0.902, n=40). Similar trends were observed for cobalt and zinc. This increase in trace metal concentrations further magnifies the problems of acid drainage. The slopes indicate that as pH decreases one unit, copper and nickel concentrations will increase by about an order of magnitude.

Comparison of Laboratory and Field Data

Laboratory data on the test stockpile solids suggested that the smaller particles had a marked influence on the stockpile drainage quality. Particles smaller than 2.0 mm in diameter comprised about 12 percent of the total stockpile mass. The weighted average sulfur content of these particles was 1.0 percent as compared with 0.6 percent S for the bulk rock. The sulfur content of these particles tended to increase as particle size decreased, with particles of diameter less than 0.053 mm having a sulfur content of 1.86 percent (Lapakko et al., 1986).

The pH range generated in the laboratory was quite close to that observed for the test stockpile drainage in 1982, when the solids were collected (figure 5). Comparison with field data from subsequent years is somewhat tenuous since the stockpile was dismantled, and the rock was divided among six different test plots. The diameter of the particles used in the laboratory tests was in the range of 0.053 to 0.147 mm, and particles in this size range comprised only about three percent of the stockpile mass. The sulfur content of the

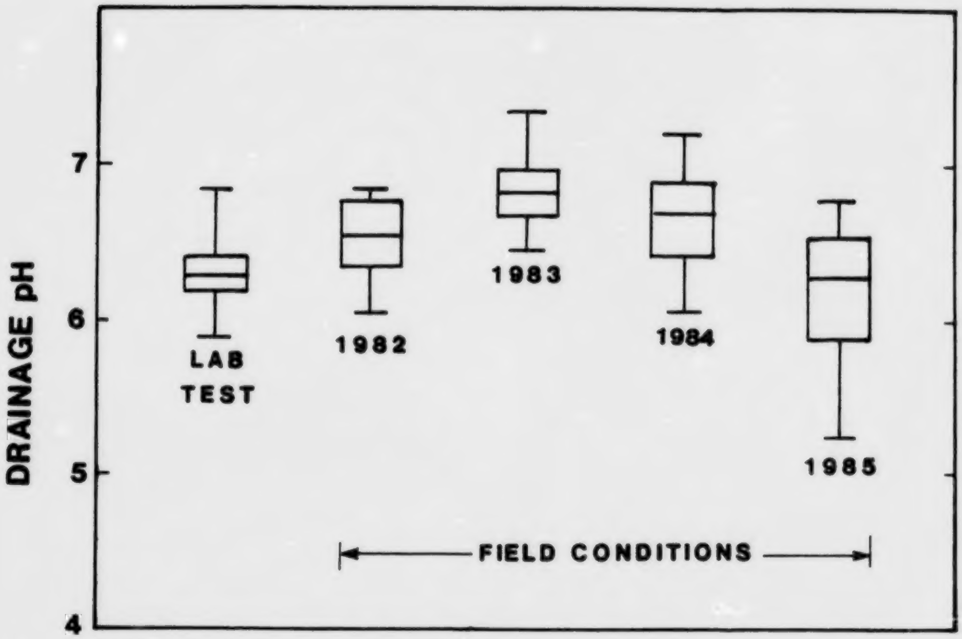

Figure 5. Drainage pH in laboratory compared to field values. Note: In September 1982 the test pile from which these laboratory solids were taken was dismantled and placed into test plots. Field data subsequent to 1982 is from these plots.

particles was 1.26 percent, as compared with 0.6 percent for the bulk rock. Since the laboratory pH was slightly lower than that observed in the field, the particle size range dominating the stockpile drainage chemistry may be slightly larger than that used in the laboratory.

Particle size is important for several reasons. As previously mentioned, sulfur content increased as particle size decreased. Sulfide minerals are softer than the silicate minerals and, therefore, are more prone to breakdown. Consequently they tend to concentrate in the smaller size fractions. Secondly, the surface area of the fines is relatively high since specific surface area increases as particle size decreases. Since the rate of iron sulfide oxidation was proportional to the available surface area, the rate of sulfide oxidation per unit mass of these particles is magnified.

Thirdly, sulfide minerals in the small size fractions are most likely to be free rather than occurring as interstitial or included minerals. For Duluth Complex samples, the diameter at which sulfides are liberated from the host rock has been estimated as 0.074 mm (Weiblen and Morey, 1976); mineral beneficiation studies suggest a value of about 0.2 mm (Vifian and Iwasaki, 1968). In particles larger than this critical size, a large fraction of the total sulfide surface area would be protected by the host rock, and therefore unavailable for reaction. Leaching tests run with larger particles would tend to yield lower rates of sulfide oxidation and acid production. For example, the leaching of minus 10 mesh (diameter less than 2 mm) tailings containing 0.92 percent S produced neutral leachate (Natarajan and Iwasaki, 1983), while the 0.053 to 0.147 mm solids of the same sulfur content produced acidic drainage in the present study. For reasons discussed earlier, the submerged reaction environment in the shake flask leaching of tailings may also have increased the efficiency of buffering by permitting acid contact with a larger silicate surface area.

The pH of drainage from test stockpiles also decreased as the sulfur content increased. The most recent annual median pH values from piles containing 0.6 percent S were in the range of 5.25 to 6.5. The corresponding pH values for stockpiles with sulfur contents of 0.8 and 1.4 percent were about 4.25 and 3.6, respectively. These values suggest that stockpiles containing 0.6 percent S are near the boundary of acid production. In the laboratory this boundary occurred between 0.80 and 0.92 percent S. This indicates that the laboratory tests simulate the leaching behavior of stockpiled solids having a bulk sulfur content roughly 68 to 79 percent of the laboratory solids. This "effective sulfur content" of stockpiled rock will, however, vary depending on the particle size distribution of the stockpiled solids. Additional study on the effects of both composition and particle size on drainage quality are required in order to more accurately extrapolate laboratory results to stockpiles in the field.

The laboratory data also closely simulated other aspects of stockpile drainage chemistry, indicating that the two reaction regimes were similar. The variation of pH over time was similar in both cases. The drainage pH in the field tended to decrease over time, reflecting the decrease in buffering by silicate mineral dissolution. For the stockpiles containing 0.6 percent S the pH decrease over time was gradual, but for the higher sulfur piles the decrease was

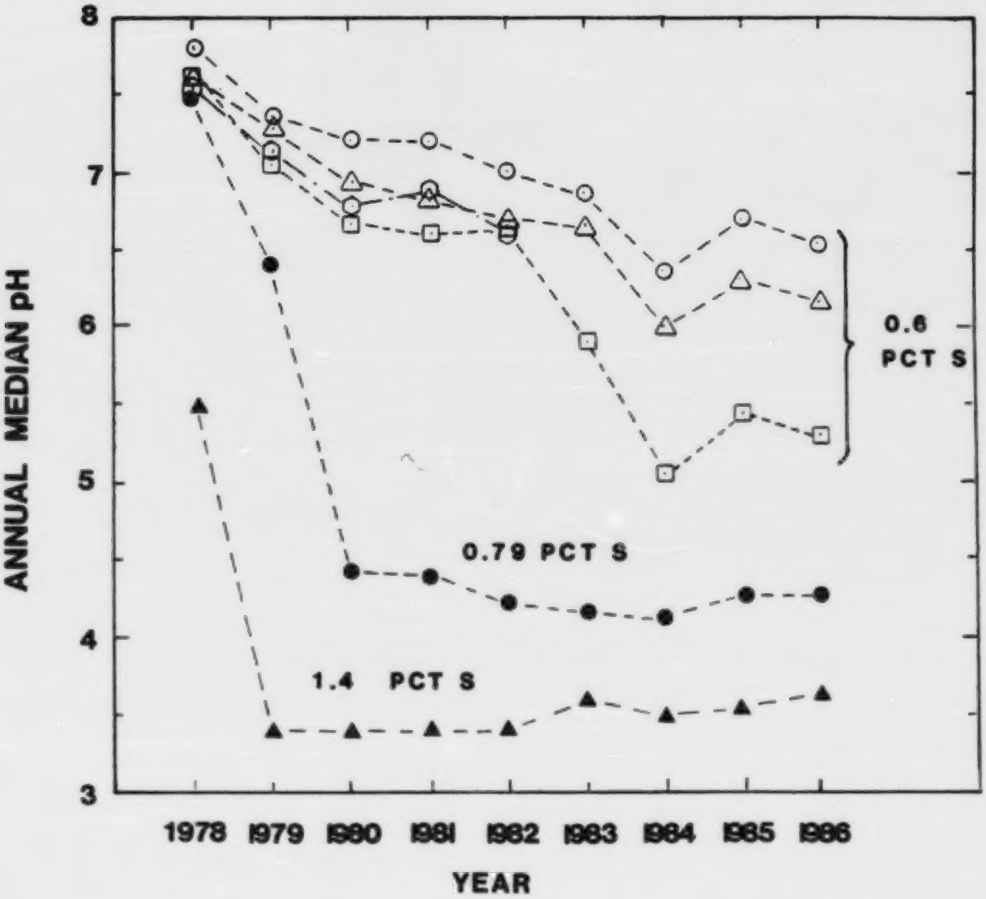

Figure 6. Annual median pH values of test stockpile drainage from 1978 to 1986 (modified from Eger and Lapakko, 1985).

relatively rapid (figure 6). After the initial pH decrease with the high sulfur piles, the drainage pH values were fairly stable. With the high sulfur samples in the laboratory tests, the more rapid pH decrease and subsequent period of relatively stable pH were also observed. The ratio of the sum of calcium and magnesium release to sulfate release traced pH fluctuations in the field as in the laboratory.

Metals concentrations increased as pH decreased in both laboratory and field settings. The slopes for the log metal versus pH relationships, however, were steeper in the laboratory than in the field. As previously mentioned, the laboratory data for the 0.92 percent solids indicated slopes of -1.11 for copper and -0.99 for nickel. Using the annual median values for metal concentration and pH for all stockpile drainages, the slopes for copper and nickel were -0.76 and -0.66, respectively (Eger and Lapakko, 1985).

Conclusions

Ten Duluth Complex samples, obtained from core drilled by AMAX exploration, were leached in a laboratory experiment. The number of samples was extremely small relative to the vast extent of the Duluth Complex. This experiment represents the starting point for examination of a wider compositional range, as well as the influence of other variables on the dissolution reactions.

The sulfur content of the samples examined ranged from 0.47 to 2.17 percent, the variation due largely to variable content of iron sulfides, in the form of pyrrhotite. Solids with sulfur contents of 0.8 percent or less produced neutral drainage while, with one exception, solids containing at least 0.92 percent S produced acidic drainage. The rate of iron sulfide oxidation, and the attendant acid production, increased linearly with sulfur content, indicating a first order dependence on available sulfide surface area. The rate of acid neutralization, which was controlled by silicate mineral dissolution, was relatively constant among the samples despite variations in the relative amounts of plagioclase, olivine, and pyroxenes present. This uniformity in neutralization rates adds credence to the possibility of predicting acid producing potential based on the sulfur content of Duluth Complex mining wastes, since they will be comprised primarily of these silicate minerals. At the 0.92 percent sulfur content, the rate of acid production due to iron sulfide oxidation exceeded the dissolution rate of silicate minerals which comprised 80 to 92 percent of the solid phase.

A solid containing 1.17 percent S did not produce acidic drainage since it contained about three percent calcium carbonate, or calcite, which is rare in the Duluth Complex (Stevenson et al., 1979). Unlike the silicate minerals, the calcite was able to neutralize the acid produced by the iron sulfide oxidation. The influence of this small mineralogical anomaly on drainage quality emphasizes the importance of examining the leaching behavior of numerous samples. Only then can leaching be described in terms of more general characteristics, such as sulfur content.

The laboratory leaching results were consistent with field observations on the variation of pH with respect to percent sulfur and time. The range of 0.8 to 0.92 percent sulfur in the laboratory appeared to be similar to a bulk sulfur content of approximately 0.6 percent for stockpiled rock. The drainage chemistry of stockpiles appears to be strongly influenced by the small particle size fraction, in which the softer sulfide minerals are concentrated. In conjunction with an elevated specific surface area, this produces a high sulfide surface area available for chemical reaction.

The equilibration period for the drainage pH is variable, and the experimental time frame must allow for this equilibrium to be reached. Under field conditions, the drainage pH from three gabbro stockpiles containing 0.6 percent S has decreased for nine years, indicating that equilibrium has not yet been reached. Drainage pH from a stockpile containing 0.8 percent S approached apparent equilibrium after two years, while about one year was required for 1.4 percent S rock (figure 6).

Trace metal concentrations increased as pH decreased, although the pH dependence was not as strong as that observed in the laboratory. Projections based on the first year of drainage from these stockpiles would have underestimated subsequent release of acid and trace metals by at least two orders of magnitude (Eger and Lapakko, 1985). This underlines the importance of allowing adequate reaction time for equilibrium to be approached.

In the laboratory experiments, some of the acid producing materials were identified after one week, while drainage from others remained near neutral for as long as 37 weeks. The disadvantage of such long experimental runs is offset by the fact that many different solids can be examined at one time. The reaction system used was small and inexpensive. The weekly time requirement is also small. Since there are no mechanical parts in the system, it is essentially maintenance free. To leach 50 samples, and analyze for pH and specific conductance, requires approximately 4 hours per week for a laboratory technician. The time required increases slightly if samples are collected for analysis of other parameters.

This experimental design has several additional advantages. First, the extended duration allows increased accuracy in projecting the leaching behavior of solids. Second, the factors controlling the drainage chemistry can be examined by analyzing drainage samples, over time, for parameters beyond pH and specific conductance. This information may provide insight into potentially effective mitigation techniques. Third, this design can also be used to examine certain mitigation techniques. For example, the mitigative potential of alkaline addition to marginally acid producing rock from the Duluth Complex is presently being examined using this technique. Fourth, it combines desireable features from batch reactor and column leaching techniques. The reaction environment permits wet-dry cycling, so it is more representative of stockpiles and unsubmerged mine walls than that in batch reactor tests. It requires a smaller mass of material, only 75 grams in this experiment, than is often used in column tests, and has no complications with respect to constant flow rates or uniform flow distribution.

The low solids requirement also permits the examination of drill core samples, as was done in this study, without seriously depleting the core collection. This allows experimentation to begin prior to the actual mine development. Knowing in advance the leaching potential of various rock types allows for more efficient development.

For example rock with a high acid producing potential could be isolated for more intensive treatment. Small particle size material, due to its high specific surface area and elevated sulfur content, would be one candidate for intensive treatment. Segregation of these particles may be possible without extensive modification in operations which use conveyors to transport waste rock. Trace metal recovery from the small particles may be feasible, since the trace metals are concentrated in this fraction and no energy would be required for crushing. The associated iron sulfides would then be disposed with the tailings, which are of lower sulfur content than the waste rock. Secondly, this premining information would permit the factoring of mitigation costs into the economics of development. For example, a high environmental cost might preclude the mining of marginally economic areas of ore deposits.

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MATERIALS HANDLING
FOR
MINE SPOILS AND COAL REFUSE

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Material handling of coal mine overburdens is an integral part of the mining plan. Proper placement of acid-producing and calcareous strata in relation to each other may be the key to successful resolution of potential acid-mine drainage (AMD) problems. Successful techniques which include blending, segregation, total removal, and chemical treatment have been employed to minimize and, in many cases, completely preclude the AMD threat. A case study will demonstrate the use of these techniques and show how a viable mining scenario can be developed which will prevent the occurrence of AMD.

INTRODUCTION

After adequate geochemical data have been gathered from a proposed minesite, and detailed field and laboratory evaluations have been completed, mine personnel can then develop material handling scenarios specifically for their site. There are numerous parameters that must be considered when special material handling techniques are required. These will include site suitability, proposed equipment layout, blasting techniques, surface and groundwater hydrology, overburden geochemical properties, and possibly others. Based on this information, mine planners can make valid decisions as to the type of material handling techniques that are needed in order to circumvent the potential problems associated with acid generation.

There are four basic techniques which have been demonstrated successfully in the Appalachian Coal Region. These techniques are blending or ad-mixing, segregation, chemical treatment, and total removal. Briefly, blending is accomplished by ad-mixing potential acid-producing and calcareous materials and blending them using a bulldozer or other appropriate equipment. Segregation is the selective burial of acid-producing materials such that they are isolated, buried and sealed from air and water. Chemical treatment entails the addition of various chemical additives in order to deter AMD formation, and total removal involves the extraction and transportation of acid-producing materials to a pre-determined disposal site. These techniques vary greatly in application and details in execution usually control their effectiveness. A more extensive discussion of each technique is presented below.

BLENDING

Blending is the process of mixing potential acid-producing materials with calcareous materials (excess $CaCO_3$ equivalent) in order to neutralize the generation of any possible acid. The concept of blending (sometimes called mixing, co-mingling, or ad-mixing) overburden strata is the most basic and simplistic method to deter acid generation. This technique has been used successfully for years by accident rather than design. Simple though it may seem, certain factors must exist which cause the system to function properly. These functions are:

- Abundance of neutralizing (calcareous) material versus potential acid-producing materials
- Excess neutralizing materials should have free carbonates (fizz) which are readily available for neutralization of acids
- Adequate blasting, (only if necessary) or ripping techniques that generate smal, particle size fractions for proper blends
- Equipment which can manipulate large vertical zones or rock material such that blending occurs as a natural process

Implementation Procedures

1. Determine through laboratory analysis if acid-producing and/or neutralizing materials are present and exactly where they are present.
2. If present, determine in what proportion, and if they are free carbonates.
3. If neutralizing materials are in excess of 3:1 (chemically) over acid-producing materials, or if acid-producing materials comprise less than 10-15 percent (volume) of the total section, then blending may be the most viable alternative.
4. If blasting is necessary, select blasting techniques to maximize selected blends of overburden material.
5. Select equipment that can provide intended blending procedures.

SEGREGATION

Segregation is the method by which acid-producing overburden materials are selectively singled out for special burial or handling techniques. This method involves special care to detail and maintaining close control of the material handling sequences during mining. Segregation is normally used where there are insufficient neutralizing materials to recommend blending techniques or where acid-producing materials are significantly abundant in volume and/or have a high degree of acid-generating potential. This is usually the case for the burial of cleaning plant refuse. Segregation differs from layering in that acid-producing materials are isolated in one sealed zone, rather than a number of alternating layers.

Segregation of acid-producing materials sounds complex, but in most situations this procedure can be easily incorporated into the material handling plan utilizing existing on-site equipment. Factors which influence this type of burial technique are:

- The lack of available neutralizing materials that would preclude the use of blending as the primary burial technique
- The availability of sufficient clay size or fine grain materials for use as sealant to be placed over and/or under the acid-producing material

- Diversion of sub-surface waters around and away from the area where acid-producing materials are to be buried

- Burial of acid-producing materials deep enough in the backfill in order to minimize air and water infiltration

- Minimize exposure of acid-producing materials with burial as soon as possible. These materials should be covered immediately upon placement and should not be exposed if they have a pH of less than 4.0

Implementation Procedures

1. Determine through laboratory procedures the abundance and relative proportions of neutralizing to acid-producing materials.

2. Determine if blending can be accomplished; if not, plan to segregate materials.

3. Depending upon the location and extent of acid-producing material, initiate the following procedures for the preparation of the burial site for the acid-producing material:

    a. Divert all sub-surface waters from the burial site

    b. Construct a pad of non-toxic and fine-grained materials as the pavement of the burial site, if needed

    c. Place acid-producing materials into site and cover immediately with the best available, usually fine-grained, calcareous mudstones or materials with excess neutralizers

    d. Calcareous materials which are used for covering the acid-producing strata should be compacted by rubber-tired vehicles or similar techniques in order to minimize the infiltration of air and water into the toxic zone. This technique is sometimes called encapsulation or sandwiching of the acid-producing material

e. Bury acid-producing material as deep into the backfill as possible, at least 5 meters from the surface, preferably below the weathered zone

f. Divert all surface water away from areas where acid-producing materials are buried

g. Minimize infiltration by the incorporation of vegetation scheme that will use nearly all available water, such as a heavy grass/legume mixture

4. If the area where the segregated materials are buried is going to be inundated by water after burial, then only 3.c above applies. However, it is imperative that the buried materials are not in the "zone of fluctuation"; that is, they must be below the water table at all times, and not in the zone where the water table fluctuates.

Segregation has been demonstrated to work successfully throughout the eastern United States. This technique is used primarily in conjunction with the burial of coal cleaning plant refuse.

CHEMICAL TREATMENT

Chemical treatment is defined as the process where acid-producing materials are isolated and treated with chemical additives in order to neutralize the potential for acid-generation. Materials such as liming agents, agricultural limestone, hydrated lime, and burnt lime have been used extensively throughout the eastern United States to treat acid spoils and to deter the production of acid.

Currently, rock phosphate is being experimented with as an acid deterrent. Techniques such as liming the pit floor, liming the backfill, and liming the final surface have all demonstrated varied success and in most cases failures are due to inadequate application rates. However, success has been demonstrated where extremely large amounts of these agents have been applied to the pit floor and acid-generating spoils to seal off pyritic bearing zones from the air and water.

Other additives such as detergents and chemical polymers are presently being evaluated and have also demonstrated mixed success. The use of sodium laurel sulfate (SLS) in recent years has been most promising in the deterrence of bacterial activity in the acid-producing material. These techniques are extremely expensive and should be considered on a site specific basis.

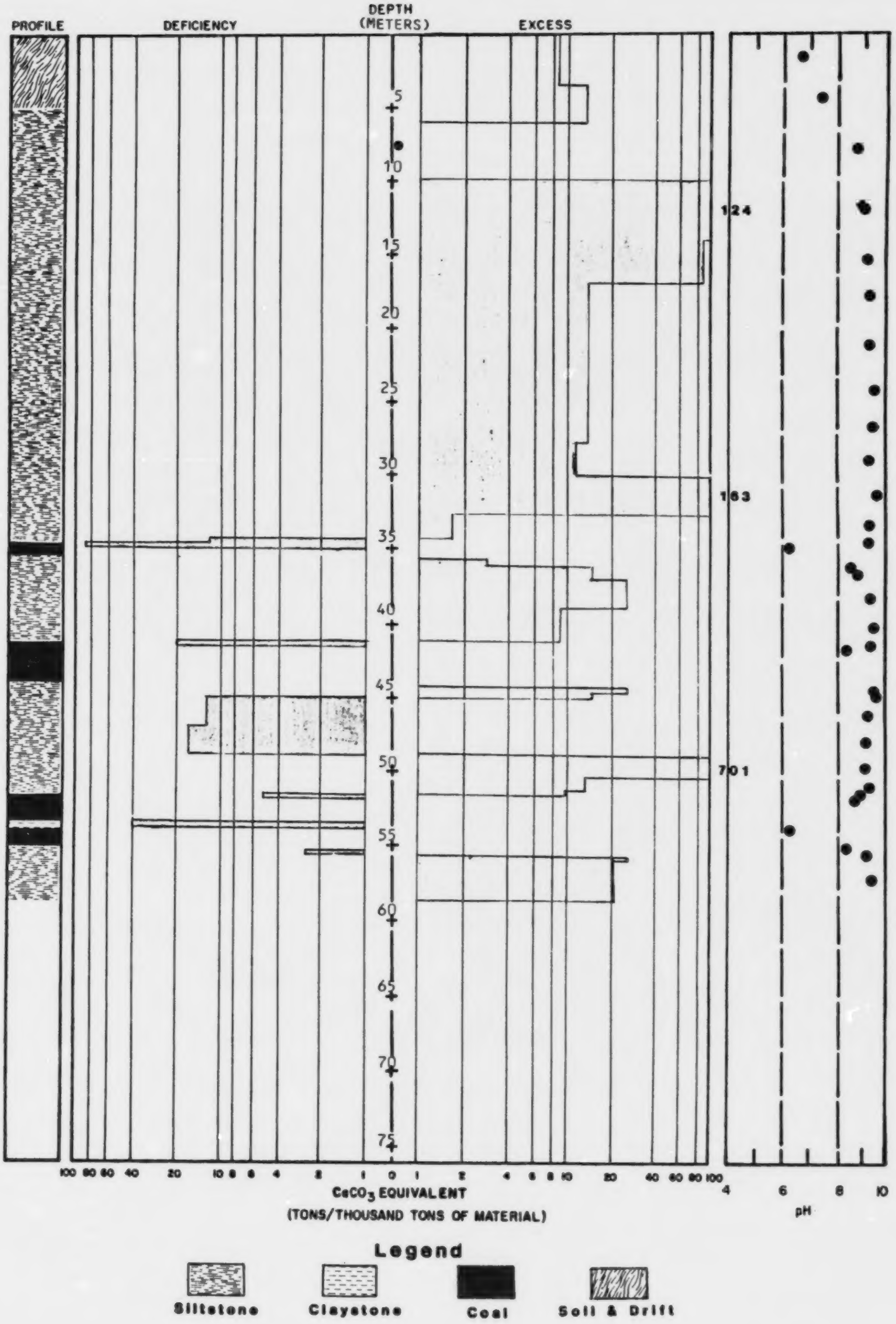
TOTAL REMOVAL

Total removal is defined as the method which requires the extraction of the entire acid-producing fraction of the overburden and the subsequent relocation to a predetermined disposal site. This method eliminates the possibility of acid being produced a the mine site. The method is very expensive if the acid-bearing zone is large; however, where the acid-bearing zone is very thin, this method could be cost effective.

In many mines in the eastern United States, where the acid-producing materials are in the form of a parting or binder, the entire coal seam and partings are removed and taken to a cleaning plant where the acid-bearing strata are removed and taken to a coal refuse disposal site where they are treated and handled properly. Generally, these refuse sites are well maintained and have constant supervision. Most disposal sites have treatment facilities which can take care of acid-waters if they should occur.

PROPOSED HANDLING TECHNIQUES - A CASE STUDY

There are four major categories of strata that exhibit high pyritic sulfur content. In descending order of magnitude, they are the hanging-wall or roof of coal, coal parting, foot-wall or pavement sections of the coal and in some cases, an acid-producing zone may be totally removed from the coal and located throughout the overburden horizon. The attached Acid-Base Account profile is very typical of many mine overburden profiles. The various units of acid-producing strata will have to be individually handled during the mining operation.



ACID-BASE ACCOUNT

The highest pyrite levels are found in the hanging-walls of the coal seams. These zones are from .1-.3 meters in thickness and pose the greatest potential for acid-generation. These zones can be handled either by blending or segregation because of the vast amounts of carbonate rock within the system. They could be buried in the adjacent pit or stored (covered) in a waste dump site. The key is to minimize exposure time to prevent oxidation and leaching, and to cover the wastes immediately with excess carbonate rich rock. Additionally, all ground and surface waters should be diverted away from these buried, potentially toxic materials.

The parting materials in the coals, which are mostly thin zones of high carbolothic mudstones, will be removed during coal loading. These thin zones will be loaded out with the coal and taken to the preparation plant. Upon removal from the coal these materials will either be stored in the slurry ponds or returned to the pits or waste dump areas. The fines which are retained in the slurry ponds should pose little or no threat because they will be inundated with water and unable to oxidize to form strong acids. The coarse refuse which will be returned to the mine dumps and pits will be moderately concentrated with pyritic minerals. These wastes should be stored and covered immediately in the same manner as the hanging-wall materials. The exposure time to air and water should be as short as possible.

The foot-wall sample from the middle coal seam (45 to 50 meters) represents five meters of potentially acid-producing material. This material has a deficiency of only 15 tonnes per 1000 tonnes and should be blended with the upper horizon material from 10-35 meters. The upper horizon materials are highly calcareous and have free carbonates and high pH's. The amount of acid-producing material is not exceptionally high and these zones should not pose a major threat if they are blended into the backfill with excess carbonate materials. The foot-wall of the lowest seam is slightly deficient and should be covered with excess carbonate material which has a fizz. This will ensure that the pit floors will be covered with a blanket of alkaline material and maintain a high pH. If these simple procedures are carried out in a disciplined manner, there should be little or no problems associated with the acid-producing strata. It is imperative however, that the mine operator minimize exposure time of the acid-producing materials and use compaction as an additional measure to ensure that air and water are deprived from the system. As mining progresses, the material handling plan should be upgraded to incorporate any changes in the geochemical nature of the overburden as may occur. Success of these procedures are not in the material handling plan itself but in its proper implementation and continual maintenance

A REVIEW OF THE PHYSICAL, CHEMICAL AND BIOLOGICAL MEASURES TO PREVENT ACID MINE DRAINAGE: AN APPLICATION TO THE PYRITIC HALIFAX SHALES

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ABSTRACT

Over the past twenty years many physical, chemical and biological approaches to reduce the acid mine drainage from mine tailings and waste rock have been tested.

Physical treatment has included tailings surface revegetation, soil amendments to the surfaces followed by revegetation, application of physical covers, such as polyethylene, asphalt, or clay covers. Also, underwater disposal of mine wastes is currently being tested.

Chemical treatment such as the application of bacteriocides has been limited but recently, the use of surfactants and controlled release surfactants have been tested.

The biological approaches tested for the reduction of acid mine drainage include the development of wetland vegetation on tailings and the use of predators toward the thiobacilli.

This report reviews various treatment approaches and develops the rational for the selection of clay capping of the waste pyritic shale pile at the Halifax International Airport.

Keywords: acid mine drainage, Thiobacilli, *T. ferrooxidans*, tailings

Introduction

During construction of the Halifax Airport substantial quantities of pyritic shale were excavated and stockpiled on the airport property. On weathering, this material produced an acidic drainage which contains numerous metallic ions in solution, analogous to the drainage observed in mining environments. Mining of base metals and coal result in an influx of inorganic sulphur compounds into the environment. Once the sulphide materials are exposed to oxygen the sulphide minerals undergo both chemical and biological oxidation. Waste dumps associated with base metal, uranium or coal mining operations have the potential of producing as much as 300 kg of sulphuric acid per ton of waste material (assuming a 10% pyritic sulphur content).

Almost since its isolation in 1947, *T. ferrooxidans* has been regarded as a possible agent in the problem of acid mine water generation, although it was first thought that it merely accelerated the precipitation of $Fe(OH)_3$ (Colmer and Hinkle, 1947). Later experiments (Leathen, Braley and McIntyre, 1953) suggested that *T. ferrooxidans* also increased the rate of pyrite oxidation. In 1970, Singer and Stumm demonstrated that direct oxidation of pyrite by air was too slow to generate the amount of acidity observed in nature and suggested that *T. ferrooxidans* acts as a necessary catalyst for the relatively more rapid oxidation of pyrite by aqueous ferric iron. The biologically catalyzed reaction was determined to be 10^6 times more rapid that straight chemical oxidation (Singer and Stumm 1969, 1970).

T. ferrooxidans is an autotroph, i.e. it obtains all of its cellular carbon by fixing atmospheric CO_2 and does not require organic substrates. It is a strict aerobe and uses molecular oxygen to oxidize sulphides to sulphate and ferrous iron to ferric iron, and it derives all of its cellular energy from these reactions.

As *T. ferrooxidans* require both atmospheric oxygen and water for growth, most acid mine water abatement studies have attempted to limit or prevent contact between atmospheric oxygen and water with sulphide minerals in the mine wastes. Various techniques to prevent acid mine drainage have been tested and these are briefly outlined below:

Physical Methods of Mine Waste Treatment

I Direct surface revegetation.

In 1971 CANMET initiated research on the direct seeding of vegetative covers to the surface of the course fraction in uranium tailings ponds.

Research began with laboratory studies of the physical and chemical properties of the tailings, followed by vegetation growth tests in environmental chambers and finally field trials. These studies led to the development of an effective and practical vegetation scheme, estimated to cost from $4,000 to $6,000 per hectare.

The procedure for establishing a vegetative cover on uranium tailings is summarized in Table 1. It involves an initial treatment to neutralize acidity in the intended root zone and to provide a basic load of nutrients. This is followed by cultivation and seeding with a mixture of two grasses. Fertilizer is subsequently applied monthly during the growing season for a five-year period after which the vegetative cover is considered self sustaining. (Moffett et al. 1977).

Although this vegetation program results in a stabilized tailings surface which resists wind and water erosion and greatly improves the appearance of the tailings basin, such a treatment scheme has not improved the water quality nor reduced the volume of the acidic drainage at the test site (Moffett et al., 1977).

Table 1 - Scheme for Vegetating Uranium Tailings

Year 1	Year 2	Year 3 to 5
Clear area of - debris	Fertilize with 33-0-0 at 0.011 kg/m² (100 lb/acre)	Fertilize with 33-0-0 at 0.011 kg/m² (100 lb/acre)
Apply limestone at 2.24 to 6.72 kg/m² 10 to 30 tons/acre)*	Reseed bare patches	Fertilize with 5-20-20 at 0.022 kg/m² (200lb/acre)⁺
Fertilize initially with 5-20-20 at 0.022 kg/m² (200 lb/acre)		
Rototill top 15 to 20 cm		
Seed** and fertilize with 0-46-0 at 0.011 kg/m² (100 lb/acre)		
Fertilize with 5-20-20 at 0.022 kg/m² (200 lb/acre)		

* Rate of limestone application is specific to each tailings area.
** Seed recommended is a 40 : 60 mixture of Redtop and Creeping Red Fescue at 0.0055 kg/m² (50 lb/acre).
⁺ Apply at monthly intervals during growing season.

II Surface amendent and revegetation

Falconbridge and Inco have found that a crushed rock layer between the tailings surface and the upper layer of overburden prevents capillary migration of acid into the applied overburden which is used as a substrate for revegetation. Again, although this treatment improves the appearance and the water and wind erosion resistance of the surface, there is little or no effect on the water quality nor on the volume of the tailings effluents (R.E. Michelutti, personel commun.).

III Impervious capping

People have suggested covering tailings and waste rock disposal sites with heavy gauge plastic, mylar films and asphalt to prevent oxygen diffusion and water permeation into the tailings. Small tests have been conducted but the plastic films tear easily and the asphalt tends to develop frost heaves and fractures with extreme temperature changes. Further, the cost of such treatments are greater than what most companies consider acceptable for environmental protection (D.R. Murray, personel commun.).

IV Surface application of organic residues

Studies are in progress on the application of 10-20 cm layers of sewage sludge, compost, wood waste or wet-land vegetation to provide an oxygen absorbing organic layer on the tailings surface.

Studies within CANMET have indicated that the application of a twenty centimetre layer of wetland vegetation will reduce acid generation and metal ion leaching within the tailings. However, these results are only obtained when the tailings are completely saturated with water as acid generation is re-initiated if the water table within the tailings drops below the tailings organic layer interface. (Silver & Ritcey, 1985)

V Underwater disposal of tailings

A survey study of tailings sites in Manitoba by Acres Consultants Ltd. has provided preliminary data which indicates that tailings submersed in water are chemically and biologically inactive. CANMET has had large scale column studies underway for the past 18 months in which pyritic tailing are under about 5 m of water. To date, the results observed support the observations made in Manitoba.

VI Clay capping and revegetation

In 1985, Harris and Ritchie presented data on the clay capping of two waste rock dumps at the Rum Jungle uranium mine in the Northern Territory of Australia. The surface of these dumps which are 20 m high and have diameters of 300 m and 500 m containing $6.5 \times 10^5 m^3$ and $4 \times 10^6 m^3$ of material respectively, were recontoured for drainage and then capped. The upper surface of the dumps were covered

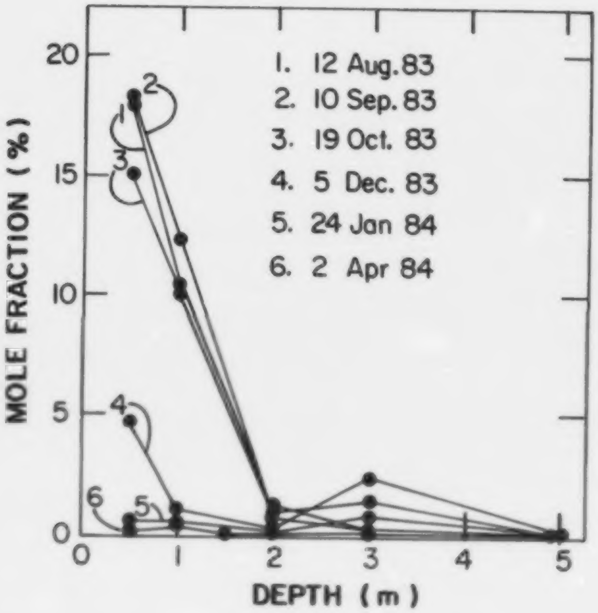

Fig. 1. Oxygen content of the clay-capped dump at various depths with time.

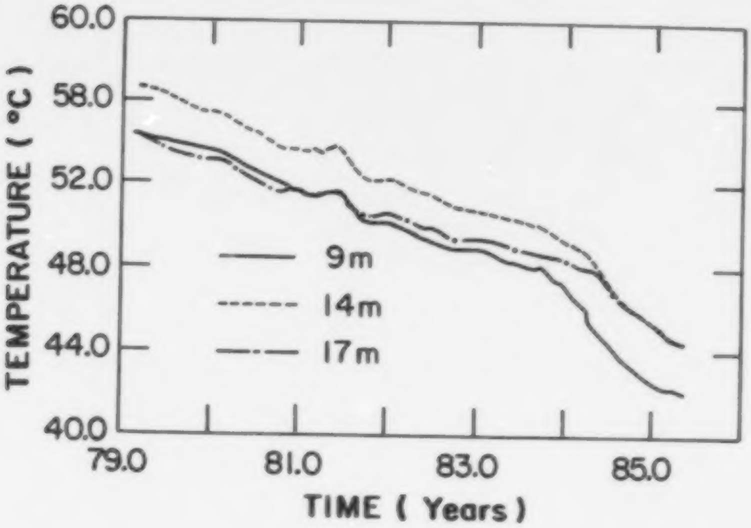

Fig. 2. Internal temperatures of the clay-capped dump with time.

with a minimum of 225 mm of clay, followed by 250 mm of clayey sand and finally with a minimum of 150 mm of gravelly clayey sand. The sides were resloped to 1:3 and covered with 300 mm of clay, 300 mm clayey sand and 150 mm of broken rock. The top surface was then revegetated to prevent erosion. Piezometers were installed and the oxygen concentration and internal dump temperatures were monitored with time. These results are shown in Figures 1 and 2.

Acid generation has been substantially reduced as the dump became anaerobic and the internal temperature declined. However, trees were allowed to invade the site and root penetration has ruptured the clay seal and the researchers are afraid that acid generation will gradually increase due to the entrance of oxygen into these dumps. The cost of capping and revegetating these two dumps was estimated at one million dollars.

Chemical Treatment of Tailings

As an alternative to physical treatment of acid mine drainage, chemical inhibition of the thiobacilli has been considered.

I Bacteriocidal agents

As tailings are disposed in the open environment, the use of bacteriocides is limited due to the potential leaching of these agents into the surrounding, normal environment where they may create further problems.

II Surfactants as bacteriostatic agents

The United States Bureau of Mines has been pursuing and testing the use of surface active agents to lyse the organisms indigenous to tailings and waste rock. The surfactants alone have a very limited effective life-time due to dilution and leaching from the piles. However, BF Goodrich and the USBM have developed slow-release rubber based surfactant pellets which are anticipated to be effective over two and one half years (Kleinmann et al. 1981, Kleinmann and Erickson, 1983).

Biological Procedures

I Predator organisms

Rotifers are known to consume Thiobacilli in large numbers as they have been found in the ditchwater utilized for bacterial leaching of copper at the Britannia Beach Mine north of Vancouver, B.C. (Oliver, 1967). More recently, the Ontario Research Foundation under contract to CANMET studied the use of zooflagellates and ciliates as predators of *T. ferrooxidans* (see figure 3).

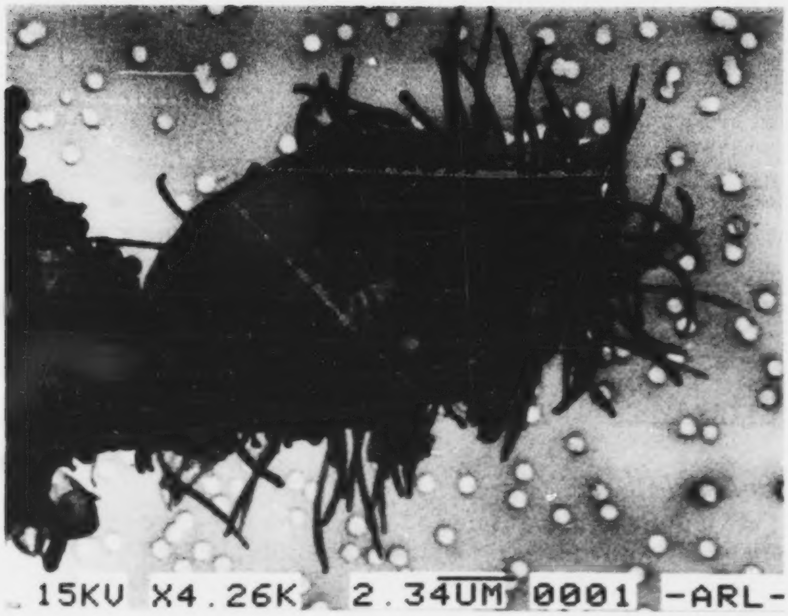

Fig. 3 Ciliate predator of _T. ferrooxidans_

Unfortunately, these predators seem to require high cell densities (> 10^6/mL) to initiate feeding and once their appetite is satisfied they tend to revert to an encysted state, allowing the thiobacillus populations to regenerate. Also, because of their relatively large size (> 25 µ θ) they are unable to enter the interstital spaces within the tailings to seek out their prey.

II In situ development of wetland vegetation

One of the more novel, and most promising, approaches to reducing acid mine drainage from tailings is the ecological engineering approach of Boojum Research Ltd. Boojum is attempting to select and propagate plants which are tolerant to the ambient conditions of a particular tailings site in order to stabilize the surface initially and over the long term to generate an organic layer which will create reducing conditions within the tailings.

Treatment of The Pyritic Wastes at the Halifax Airport

Due to construction of new facilities at the Halifax International Airport excess pyritic shale and overburden have been stockpiled on the property. On exposure to the atmosphere and precipitation the pyritic components of the shale have undergone oxidation resulting in acid generation and leaching of metals from this waste material.

Transport Canada reviewed the various methods for mitigating acid mine drainage from their waste pyritic shale pile. Since clay was locally available and in order to minimize the cost, Transport Canada decided to cap the waste pyritic shale dump with clay and an overlaying layer of soil to permit revegetation.

During studies on the reclamation of solonetzic soils, it was observed that Thiobacilli were unable to sustain viable populations in soils containing 4.8 meq of Na^+/100 g of soil. (McCready 1982; McCready and Krouse, 1982).

Later studies on the adaptation of Thiobacilli to growth in saline waters indicated that T. ferrooxidans could not tolerate sodium chloride concentrations greater than 1% w/v (McCready, unpublished).

It was therefore postulated that the incorporation of a salt layer between the pyritic shale waste and the overlaying clay cap would add an additional bacteriostatic activity to the oxygen depletion effect of the clay capping.

A laboratory study was conducted to test this hypothesis.

Materials and Methods

Four columns (6 cm dia x 50 cm length) were loaded with one kg of Halifax shale and overburden which was supplied by the Department of Transport as shown in Fig. 4.

Two of the four columns had 5 grams of road salt (NaCl) placed on the surface of the shale. One salt supplemented column and one unsupplemented shale column were inoculated with 500 mL of a 96 hour culture T. ferrooxidans grown on the low phosphate medium (McCready et al., 1986). The other salt-supplemented column and the remaining unsupplemented column were trickle leached with

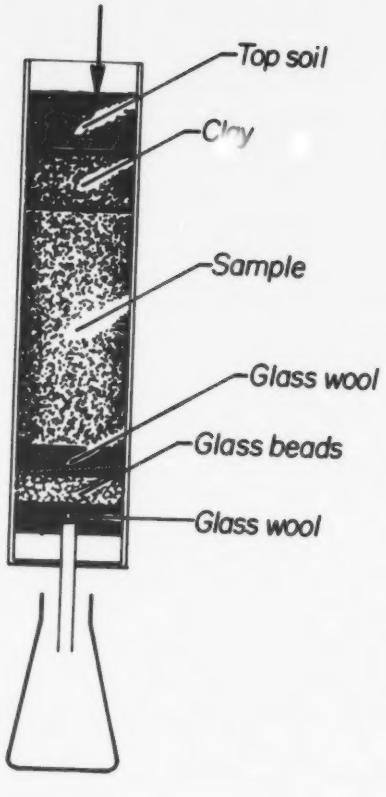
Top soil
Clay
Sample
Glass wool
Glass beads
Glass wool

Fig. 5 - Column design for testing clay-capping.

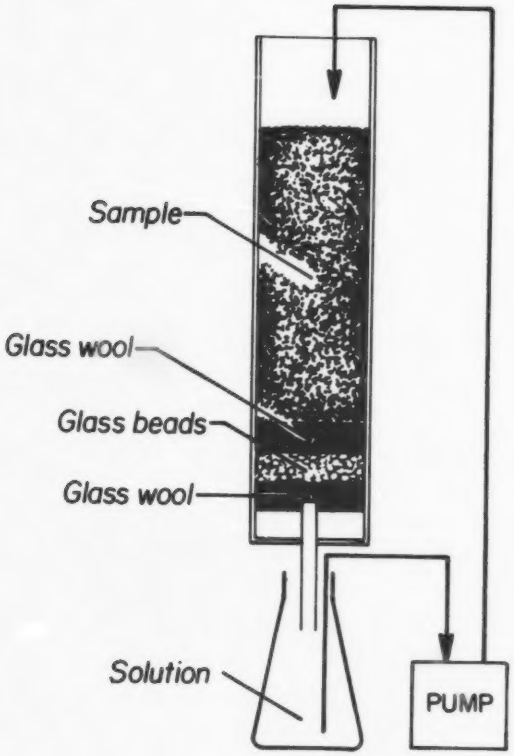
Sample
Glass wool
Glass beads
Glass wool
Solution
PUMP

Fig. 4 - Column design for testing the effect of 1.5% NaCl in solution.

distilled water. The distilled water and the *T. ferrooxidans* cultures were circulated through the columns using a Gilson Minipuls-2 peristalic pump set at a flow rate of 300 mL/h.

Samples of the recirculating solutions were collected periodically from the two columns for pH determinations and ferric iron analyses. The total volume of solution for each column was kept constant at 500 mL by the periodic addition of distilled water.

To assess the effect of a salt (NaCl) and clay cap two additional columns were prepared as shown in Figure 5. The test column had a layer of road salt placed between the pyritic shale sample and the clay layer which was tightly compacted in both columns. Two hundred and fifty mL of distilled water was added to each column to assess the percolation rate through the clay.

Results and Discussions

Effect of road salt on ferric iron production

When the cultures of *T. ferrooxidans* were recirculated through the pyritic shale columns the soluble ferric iron content of the solution decreased due to adsorption of the iron to the clays in the shale/overburden mixture. After 1 month, the soluble ferric iron concentration of the leachate increased in the nontreated column due to microbial activity (Fig. 6). This was confirmed by microscopic examination of the leachate; after 50 days the leachate contained about 10^6 organisms/mL.

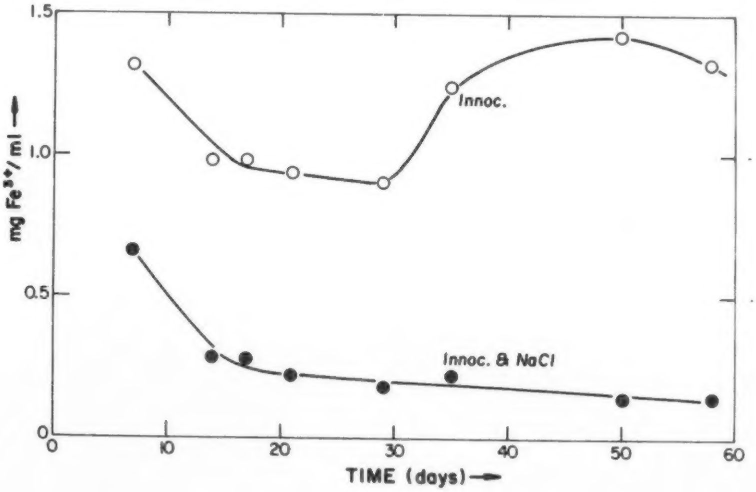

Fig. 6 - Fe^{+3} in test column effluents.

In contrast, in the salt treated columns, the soluble ferric iron content decreased over the first 20 days and remained at a minimal concentration until the experiment was terminated. No active bacterial cells were observed during microscopic examination of the leachate.

In the two columns leached with distilled water there was very little iron release and no T. ferrooxidans could be detected in the leachate. Extensive physical breakdown of clay components of the overburden were observed in the salt supplemented column. The continuous recycle of the saline solution most likely resulted in the stripping of calcium ions from the clays and the sodium ion replaced the calcium in the binding sites. This resulted in the formation of clay fines which plugged the column after 35 days. Repeated back-flushing of the column finally resulted in clearing of the silt and the experiment was continued to the 58th day.

Effect of a salt and clay capping

After compacting the clay and soil layers on the two columns, 250 mL of distilled water was placed on the top of the columns. None of this water penetrated through the clay layer in either column. After two weeks the added water had evaporated and the soil and clay layers were allowed to dry out.

The clay layer in the column without salt cracked vertically and on rewetting with 250 mL of water, the water penetrated through the clay layer and the column within 4 hours; the clay seal has not resealed to date.

In contrast, the clay and salt seal in the other column cracked horizontally on drying. When 250 mL of water was reintroduced it took 12 hours for the water to penetrate through the clay and pyritic shale. An addition of a second 250 mL aliquot of water, the clay re-sealed itself and no additional water percolation through the column has been observed to date.

The water which passed through these two columns initially was analyzed for iron content and pH. These values are presented in Table 2.

Table 2. pH and ferric iron concentration of the initial effluent and after 250 mL of effluent had passed through the clay sealed columns.

		pH	Fe^{+3} ppm
Clay alone	First 5 mL	2.47	292
	After 250 mL	2.60	82
Clay + NaCl	First 5 mL	2.41	98
	After 250 mL	2.51	34

As stated above, the clay capped column without salt added has not resealed to date. After drying and cracking with subsequent resettling of the clay, water percolation was observed to be 55 mL/day. After four months, the percolation rate has declined to 22 mL/day and there has been 0.75 increase in the effluent pH over this period (Figure 7). In contrast, the column with salt added, sealed completely after drying and the subsequent addition of 500 mLs of water, and to date no further effluent has been collected. Removal of an interstitial water sample indicated that the water has a pH of 2.98 with a very low iron content (48 ppm).

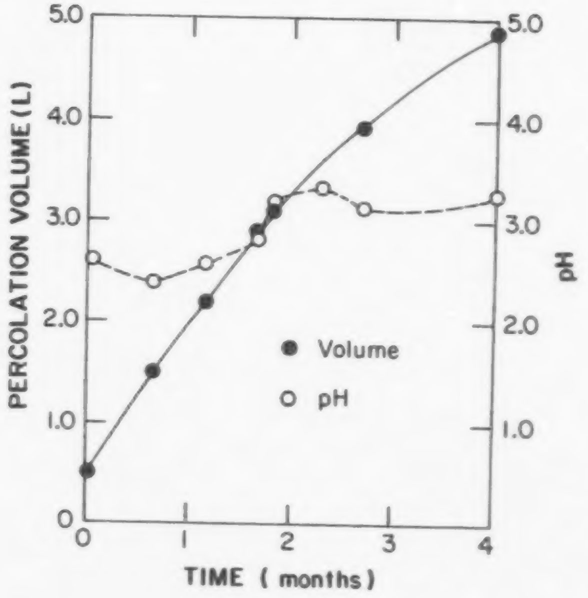

Fig. 7. Volume and pH of the effluent from the clay-capped column without the salt layer.

Conclusions

The addition of a salt layer between the pyritic shale and the clay cap resulted in: (1) a higher water retention time by the clay cap as indicated by the longer time required for drying and cracking of the clay layer. (2) On drying, the clay cap cracked horizontally, rather than vertically, thereby preventing oxygen penetration to the underlaying shale. (3) On re-wetting, the salt induced swelling of the clay results in the reformation of an impermeable seal more quickly than in the absence of salt.

In addition, if the clay cap is ruptured, any water percolation would result in the leaching of soluble salt into the shale thereby inhibiting any thiobacilli within the water penetrated shale.

These preliminary results, indicate that that application of a salt layer between the pyritic shale and the overlaying clay cap should be beneficial in maintaining the cap integrity and simultaneously provide a bacteriostatic effect within the pyritic shale. Future studies should consider the mineral composition of the clays to select the best clay(s) based on their water and salt retention capacity for use as a capping material for pyritic wastes.

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THE USE OF SURFACTANTS TO PREVENT AMD IN COAL REFUSE AND BASE METAL TAILINGS

by

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Abstract.

Thiobacilli bacteria, by direct or indirect mechanisms, oxidize metal sulfides and produce acid and soluble metals at a much faster rate than chemical oxidation. Elimination of these bacteria would inhibit acid formation. Several inorganic and organic compounds can inhibit thiobacilli growth; however, organic compounds are most effective. Anionic surfactants are some of the most effective organic compounds that are commercially available. Anionic surfactants become bactericides at low pH values. The short duration of these chemicals in the environment makes it necessary that a system such as the ProMac System, comprising a quick-acting liquid spray with a long-term controlled release pellet, be used for permanent reclamation.

INTRODUCTION

Pyritic materials and reduced metal compounds associated with coal seams and metal ores oxidize and form mineral acids when exposed to the atmosphere by earth disturbances like mining and construction. Acid mine drainage (AMD) from these drastically disturbed materials degrades surface water quality and the aquatic environment, and may contaminate the local groundwater system. The disturbed material (eg., coal refuse and waste rock) becomes acidic and biotoxic.

The acidic material erodes easily and contaminates adjacent lands. This cycle continues because erosion exposes unweathered pyritic or other sulfide material for oxidation. This oxidation process is catalyzed by the bacteria *Thiobacillus ferrooxidans*. A pH dependent three-stage mechanism of acid generation has been proposed by Kleinmann and co-authors (1981).

Inhibiting or destroying these bacteria can significantly slow the rate of acid production. Anionic surfactants, organic acids and food preservatives (Onysko et. al., 1984) act as bactericides and kill these bacteria; however, bactericides degrade with time and are lost because of leaching and run-off. To overcome this inherent short duration effectiveness of spray applications, controlled release systems were developed (Kleinmann, 1981; Fox and Rastogi, 1983).

Control of acid generation for prolonged periods greatly enhances reclamation efforts and can reduce reclamation costs by reducing the amount of topsoil needed to establish vegetation. A strong vegetative cover for three years or more can break the acid production cycle due to three natural biological processes: (1) a healthy root system competes for both oxygen and moisture with acid-producing bacteria; (2) populations of beneficial heterotrophic soil bacteria and fungi are reestablished, resulting in the formation of organic acids which are inhibitory to *T. ferrooxidans* (Tuttle et. al., 1977); (3) the action of plant root respiration and these heterotrophic bacteria increase CO_2 levels in the spoil, resulting in an unfavorable microenvironment for growth of T. ferrooxidans.

Iron Oxidation

The geochemical oxidation of pyrite may be characterized by the reactions reported by Stumm and Morgan (1981):

$$FeS_2 + 7/2\ O_2 + H_2O = Fe^{2+} + 2SO_4^{2-} + 2H^+ \qquad (1)$$

$$Fe^{2+} + 1/4\ O_2 + H^+ = Fe^{3+} + 1/2\ H_2O \qquad (2)$$

$$Fe^{3+} + 3H_2O = Fe(OH)_{3(s)} + 3H^+ \qquad (3)$$

$$FeS_2(s) + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \qquad (4)$$

After pyrite is oxidized (Eq. 1) and the pH is lowered (<5.0), ferric iron is produced extremely slowly (Eq. 2). This second reaction, however, can be accelerated by microbial catalysis to increase the overall rate of the oxidation of ferrous iron (Baker, 1975). With the generation of ferric iron, insoluble ferric hydroxide can then form under proper pH conditions (Eq. 3). Ferric iron can also be reduced further by pyrite, resulting in the generation of more acidity and ferrous iron (Eq. 4).

Beck (1968) shows that bacteria play an important role in Eq. 1 and 2 causing reaction rates to increase significantly. Microorganisms may directly attack pyrite or attack the Fe^{2+} generated in Eq. 1. Both reactions generate acidity.

Thiobacilli bacteria are indigenous to environments containing sulfides, oxygen, and acid conditions. Thiobacilli are the autotrophic microorganisms that obtain their energy from the oxidation of iron compounds. These organisms are most active at pH 2.0-5.0 (Manning and Cook, 1972). Chemical oxidation of Fe^{2+} is kinetically limited, but the oxidation rate can be accelerated as much as 500,000 times when iron-oxidizing bacteria are present. The resulting Fe^{3+} can directly oxidize metal sulfides producing more Fe^{2+} (Krauskopf, 1967).

Copper Oxidation

Copper minerals commonly found in porphry deposits consist chiefly of chalcopyrite ($CuFeS_2$) and chalcocite (Cu_2S). These minerals and covellite (CuS) are subject to chemical attack by ferric iron generated from associated pyritic minerals. Reactions governing pyrite dissolutions are enhanced by bacterial mechanisms result in product necessary to establish Eq. 2. Oxidation of copper compounds by ferric iron producing Cu^{2+} result in competitive solubility systems where

$$14Cu^{2+} + 5FeS_2 + 12H_2O = \\ 7Cu_2S + 5Fe^{2+} + 3SO_4^{2-} + 24H^+ \qquad (5)$$

Acidity generated by both systems improved the activity of chemoautotrophic thiobacilli promoting a continuation of the cycle dependent on the oxidation of ferrous iron and sulfur. Lundgren and others (1986) indicated that direct oxidation of monovalent copper by thiobacilli bacteria is supported by some researchers; however, it is very difficult to eliminate the influence of ferric iron on the process.

Uranium Oxidation

Low temperature uranium minerals found in podiform and roll front deposits occur in a hexavalent state. Oxidized U^{6+} transported by groundwaters and precipitated as either coffinite ($USi0_4.nH_2O$) or pitchblend (uranite, UO_2) can be solubilized by ferric iron.

$$UO_2 + Fe_2(SO_4)_3 = UO_2SO_4 + 2FeSO_4 \qquad (6)$$

Bacterial activity accelerating the production of ferric iron can directly cycle this reaction through both pyritic supplied ferrous iron and ferrous iron generated by the competitive solubility system established in Eq. 6.

Investigations by DiSpirito and Tuovinen (1982) proved that T. ferrooxidans can attack U^{4+} directly and oxidize it (Eq. 7) possibly using the same redox protein involved in the T. ferrooxidans oxidation of Fe^{2+} to Fe^{3+}.

$$2U^{4+} + 0_2 + 2H_2O = 2UO_2{}^{2+} + 4H^+ \qquad (7)$$

Nickel and Other Sulfide Mineral Oxidations

Nickel sulfide minerals occur in greatest concentration in ultramafic rocks and to a lesser extent as laterite deposits. In either type of deposit, iron is a common associate providing a system of oxidation and bacterial catalyzation similar to that described for copper. In mineral form, it commonly occurs as pentlandite $(Fe, Ni)_9S_8$.

The indirect mechanism for thiobacilli oxidation of metal sulfides follow Eq. 8 and Eq. 9. The reaction proceeds in Eq. 8 because Fe^{3+} is the oxidizing agent while in Eq. 9 an acid reaction with the mineral occurs (Lundgren et. al, 1986). Direct biochemical oxidation occurs with stibnite (Sb_2S_3), copper selenide (CuSe), arsenopyrite (AsFeS), and molybdenite (MoS_2).

$$MS + 4\,Fe_2(SO_4)_3 + 4H_2O = MSO_4 + 4H_2SO_4 + 8FeSO_4 \quad (8)$$

$$2MS + 2H_2SO_4 + O_2 = 2MSO_4 + 2H_2O + 2S^0 \quad (9)$$

The exact mechanisms of metal sulfides oxidation by bacteria directly attacking the metal or by bacteria attacking the associated reduced iron and sulfur is not important environmentally. The increased acidity of waste rock plus the increased solubility of heavy metals are the important results of thiobacilli actions. Other than the commercial process of metal ore leaching (Hutchins et. al., 1986), there are no situations where the uncontrolled catalytic actions of thiobacilli bacteria are desired.

LABORATORY STUDIES

Surfactants have unique properties that make them extremely useful chemicals. Anionic surfactants are good cleaners and emulsifiers, but have limited bactericidal properties (Walters, 1965). However, in an acid medium, anionic surfactants act as a bactericide. The optimum pH range is 1.9 to 2.2 with the bactericidal properties rapidly decreasing above pH 3.0 (Dychdala, 1983).

Kleinmann and Erickson (1983) reported that W. B. Hugo summarized the different ways surfactants affected microorganisms. Hugo concluded that the semi-permeable properties of a microorganism's cell walls were altered by surfactants. This view is supported by others (Schafa and Salton, 1960).

Coal Refuse

Control of bacteria through the use of chemical agents requires the evaluation of their effectiveness on different toxic mine wastes. The geochemistry, hydrology, acid generation potential, and neutralization potential of a specific site determine the bactericide that will function most effectively. Suitable laboratory tests that provide good reliable information about site characteristics in a reasonable period of time had to be developed. Two such tests are the Batch Incubation Test and the Column Leach Test (Shellhorn and Rastogi, 1984).

Figure 1 shows results of an eight-week incubation test on a coal refuse sample. Shown are the ensuing reductions in acidity as a function of time with different concentrations of the surfactant. Inhibition of acid generation ranged from 11% reduction for the 500 ppm treatment to 78% reduction for the highest treatment level. Corresponding changes in pH, Eh, specific conductance and metal ions were seen with time with increasing bactericide concentration.

Figure 2 shows the results from the column leaching test on a high acid-producing refuse using different concentrations of the bactericide. Acid reduction of over 90% was seen at the higher concentration.

Metal Ores

Sulfur fractionation was performed on all three materials to determine pyritic sulfur and resultant ferrous iron contents. Table 1 contains the sulfur form data for each material in which the levels of pyritic sulfur by decreasing amounts are copper, nickel and uranium, respectively.

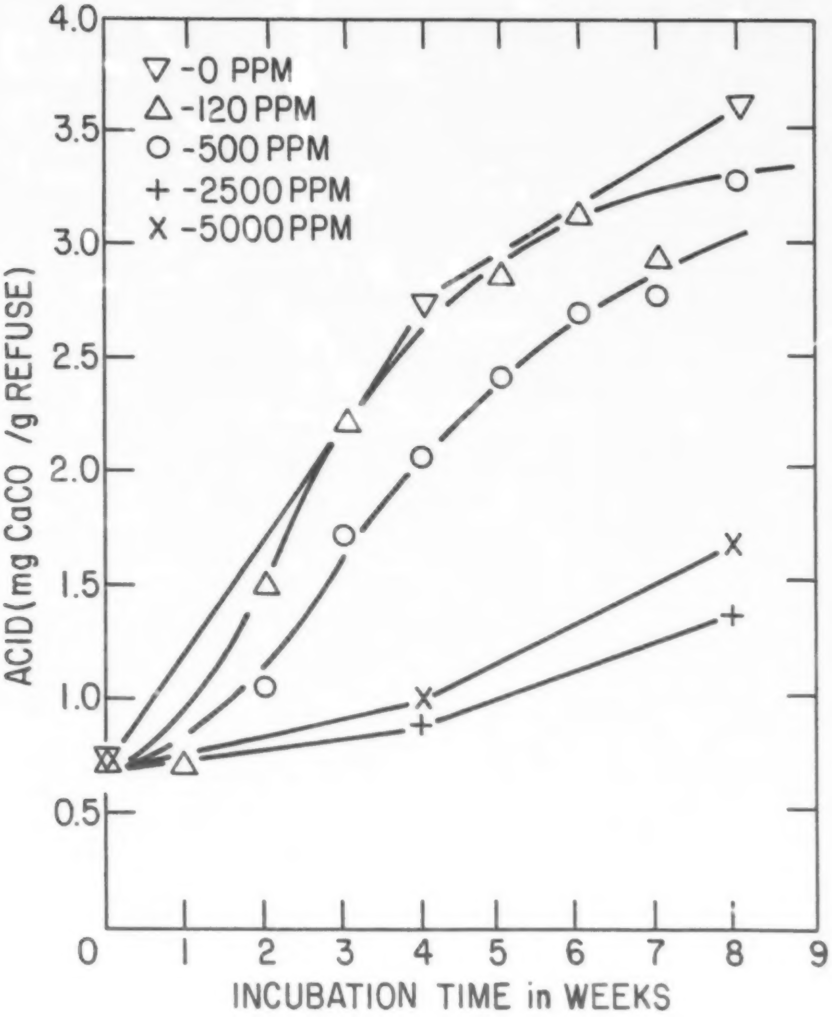


Figure 1. Incubation Test Data

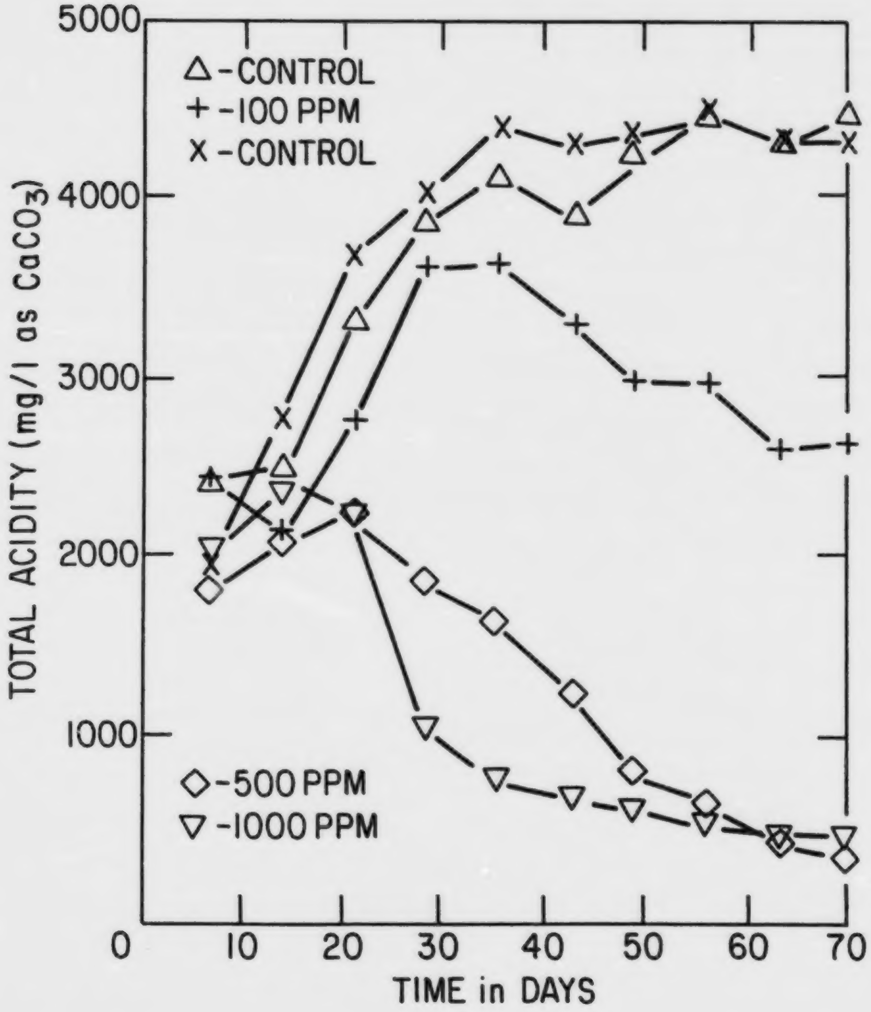


Figure 2. Column Leach Test Data

Table 1. Sulfur Fraction Determinations

Sample	Pyritic Sulfur %	Sulfate Sulfur %	Organic Sulfur %	Total Sulfur %
Copper	47.500	0.400	0.810	48.700
Nickel	5.430	2.310	3.160	10.900
Uranium	0.321	0.992	0.017	1.330

Ore tailings were extracted with water in 1:1 ratio and qualitative bacterial assessments were established by using 1:10::leachate:9K media (a ferrous iron rich medium) dilutions. Five replicates of each sample were tested, resulting in positive scores for copper and uranium leachates only. Visual observations indicated a much stronger positive response for the copper leachates to the uranium sample.

Figures 3-5 represent acidity data obtained from eight weeks of incubation testing for tailing samples of copper, uranium and nickel ores, respectivedly. As the data represent separate samples from each incubation period, the results are cumulative and should be compensated for by adjusting to a time zero baseline.

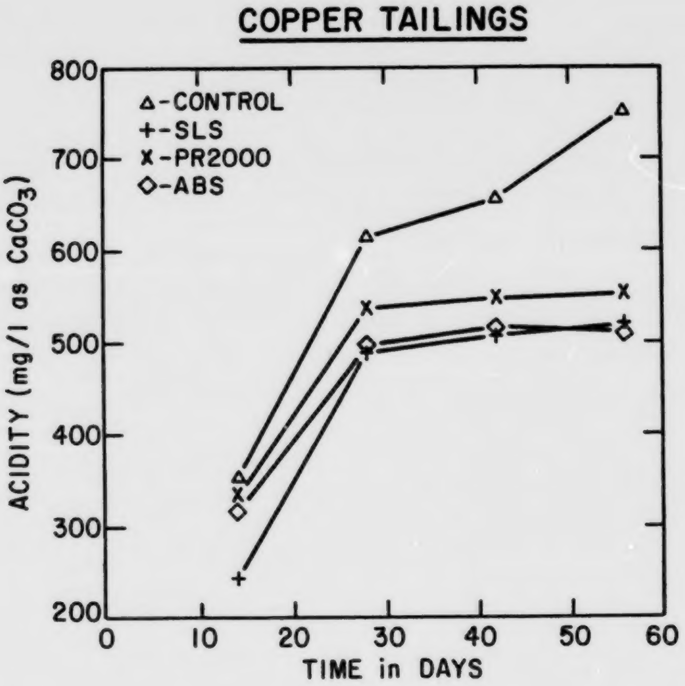


Figure 3. Copper Tailings Incubation Test

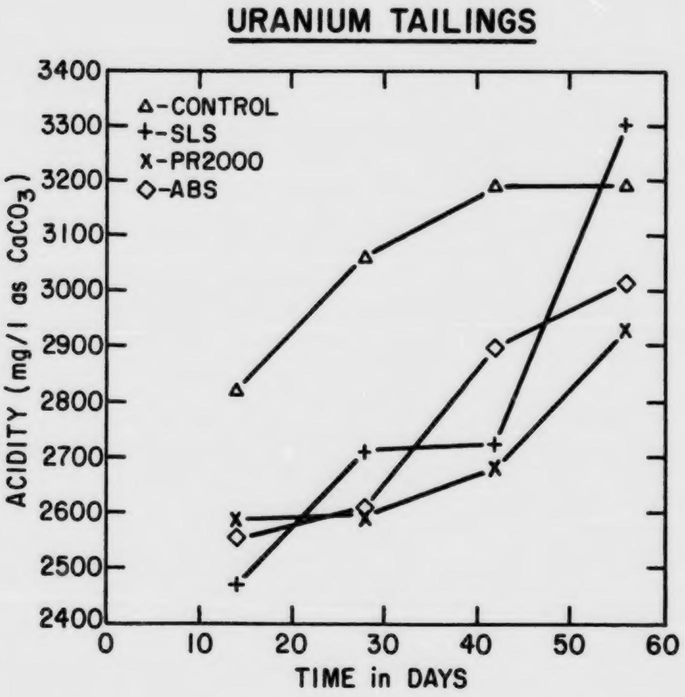


Figure 4. Uranium Tailings Incubation Test

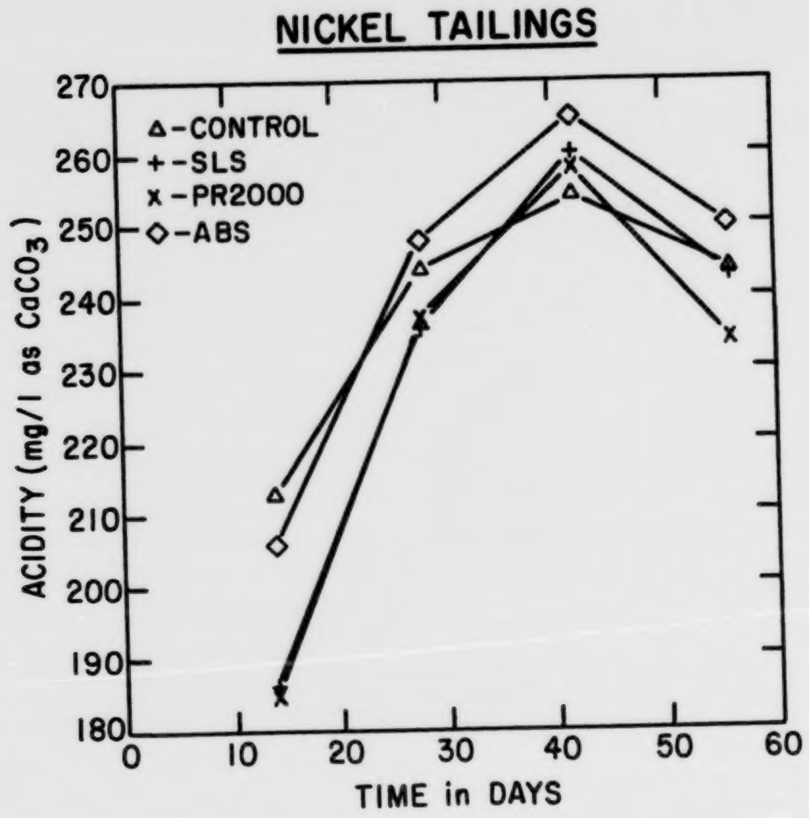


Figure 5. Nickel Tailings Incubation Test

Data trends established in Figure 3 indicate that subsequent to four weeks of incubation, acid production was minimal. A final reduction in acidity at week eight of >50% averaged for the treated samples was obtained.

Induced leachates from uranium tailings (Figure 4) suggested an immediate response to the bactericides was obtained, evidenced by 70% reduction in acid production at the second week of incubation. However, trends established through week

eight indicate a return towards control level acidity values. In contrast, results on nickel tailings (Figure 5) indicate no reductions in acid production by any of the treatments were obtained during the test period.

Acidified ferric ion solutions are effective lixiviants for many metals of sulfide and oxide ores. On this basis, the copper ore tailings possessing the highest pyritic sulfur level would be expected to produce the most acidity. Data presented in Figures 3-5 contradict this prediction, showing a significantly higher level of acidity for uranium ore samples. Nickel tailing leachates produced acidity levels at rates relative to uranium and copper tailings in agreement with pyritic sulfur analysis.

Batch incubation data by itself shows that significant inhibition of iron-oxidation and subsequent pyrite oxidation and acid formation were witnessed only for copper ore tailings. The lack of a positive bacterial response for nickel leachates suggests that inorganic kinetics were governing the pyritic oxidation. However, uranium tailing leachates exhibiting strong positive bacterial responses produced minimal levels of acid inhibition during interim incubation periods. High baseline acidities for this material may generate potential acid hydrolysis of the bactericide solutions producing limited effects. Trace metal determinations support the conclusions drawn from acidity data showing that significant ferrous-ferric solutions enhanced by acidophilic thiobacillus are prominent only in the copper and uranium tailings. Preliminary column leaching tests on uranium tailings indicate that sequential flushing removed a stored mass of oxidation products. Bactericide treated columns respond as expected showing acid production inhibited as much as 50% with a control level intermediate to that produced in copper and nickel columns. The initial results from this work are consistent with that predicted from sulfur fraction data with exception to nickel tailings. It appears that the nickel materials have not acidified to a level favorable for bacterial activity.

Summary

The batch and column leach methods provide two independent ways of evaluating bactericides and determining relative dosages required for acid inhibition in different spoils and refuse. It is important to note that in these methods no attempt was made to duplicate nature. The particle size distribution in each case was the result of artificial grinding, and in the case of columns, sieving and re-mixing different sizes to maintain uniformity. The relative surface areas exposed were very high which would lead to higher levels of acid generation and higher dosages of active agents for acid inhibition.

FIELD STUDY

Site Preparation

The Route #43 gob pile (Figure 6) near East Springfield, Ohio was regraded to provide a smooth gentle grade. A positive surface slope of approximately 10:1 was developed from the northeast edge of the pile towards the southwest. This slope was increased to a maximum of approximately 4:1 at the southwest edge.

No amendments were added to the regraded refuse pile before the ProMac System was applied. The 0.89 ha at the southern end of the site was left as a control plot while the 1.01 HA adjacent to the northwest side of the control was treated with the ProMac System.

The site, both control and treated areas, was covered with six to eight inches of topsoil. The topsoil was fertilized with 336 kg/ha of a 16-16-16 fertilizer and 6.7 metric ton/ha of lime. The area was seeded with 25 kg of seed mixture/acre followed by a 4.5 metric ton/ha of hay mulch.

Site Monitoring

Eight soil moisture samplers (pressure-vacuum lysimeters) were installed at different locations on the site. It was

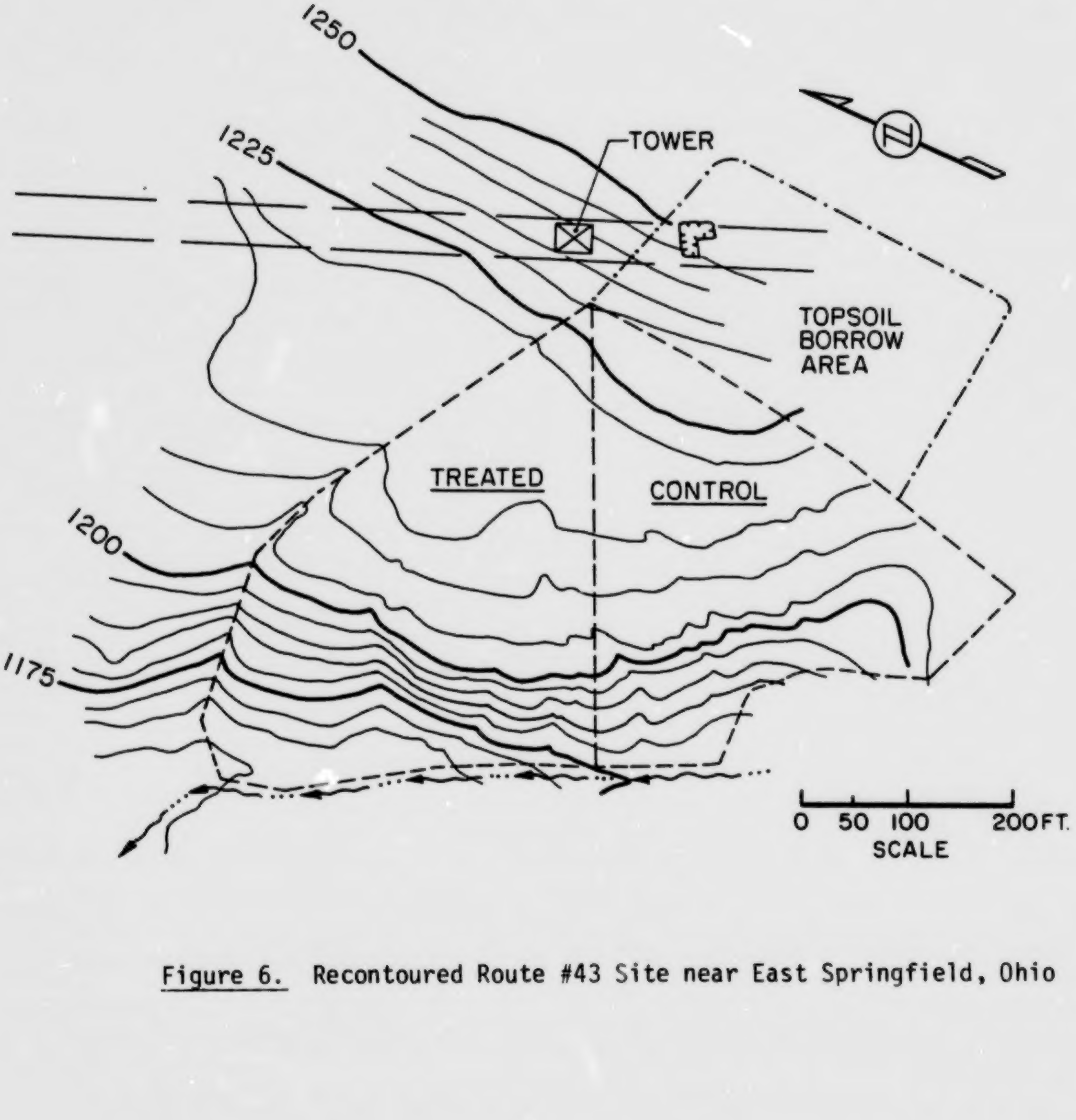

1250
1225
1200
1175
TOWER
N
TOPSOIL
BORROW
AREA
TREATED
CONTROL
0 50 100 200 FT.
SCALE

Figure 6. Recontoured Route #43 Site near East Springfield, Ohio

expected that the effects of the ProMac System application would be noticeable first in the vadose zone, and then in the drains or seeps. Actual response times are controlled by climate and the gob pile hydrology. Other instruments installed on the site were a rain gauge and an evaporation pan.

Monitoring of the site included periodic water collection and servicing of instruments as well as periodic site inspections to ascertain surface vegetation cover and acid generation. The site was photographed periodically during the past 2.5 years after vegetative cover was established.

A total of eleven locations were monitored routinely: eight soil moisture samplers, two groundwater drain pipes, and the drainage ditch flow. Monitoring was scheduled to last for at least two years or until changes were no longer observed.

Microbiology

To determine the effect that bactericide treatment would have on the microbial populations in the topsoil and upper parts of the coal refuse, a study was designed to measure the most probable numbers (MPN) of organisms that would grow in two different media. One is a semi-rich medium for the growth of general heterotrophs and the other is a specific medium for

T. ferrooxidans. This study was conducted in June 1985 and in June 1986 and will continue for a few more years.

Soil samples were taken with a soil corer from both treated and untreated sites of the pile. Each soil core had all the topsoil layer and part of the refuse material. The areas of the pile, treated and untreated, were sampled according to a grid pattern. Each area was divided into four blocks, avoiding the border zone between treated and untreated. Five samples were taken from each block, totalling 20 samples per area. Each block was combined, to yield a total of four composite samples per area. This reduced the variability between samples that originated from the same area, reduced the number of dilutions needed for testing, and enhanced the statistical significance of the test.

Three test zones, according to depth, were established: the interface between the cover soil and the refuse, the cover soil immediately above it, and the refuse material immediately below it. Each zone was 7.6 cm long and the interface zone consists of 3.8 cm of each layer.

Two separate experiments were conducted. The first was the growth of heterotrophic microorganisms in cover soil of the treated section versus cover soil from the untreated section. This comparison was conducted for composite samples of the cover soil. The second experiment was of the growth of T. ferrooxidans in treated versus untreated sections. This comparison was conducted for composite samples of the refuse layer. Each composite sample consisted of five individual cores.

Heterotrophic microorganisms were enumerated using a general heterotrophic medium with the addition of soil extract. Such medium gives rise to a large percentage of the heterotrophic population. The results after 12 months (Figure 7) are comparable to the 24-month results shown in Figure 8 (Horowitz et.al. 1985). The trends are the same, but there has been a 500-fold increase in heterotrophic bacteria numbers from June 1985 to June 1986. Higher numbers of heterotrophic microorganisms were found in the ProMac systems treated area when compared with the untreated control.

The number of heterotrophs decreased with the depth of the sample in both studies and is a result of a decrease in oxygen and available carbon source. The refuse material has an effect on the immediate cover soil. Acid generated or stored in the upper part of the refuse can reach the first few centimeters of the cover soil and prevent good heterotrophic growth. This was more pronounced in the untreated area.

Figures 7 and 8 show the results of the MPN tests for both heterotrophs (beneficial) and acidophilic bacteria

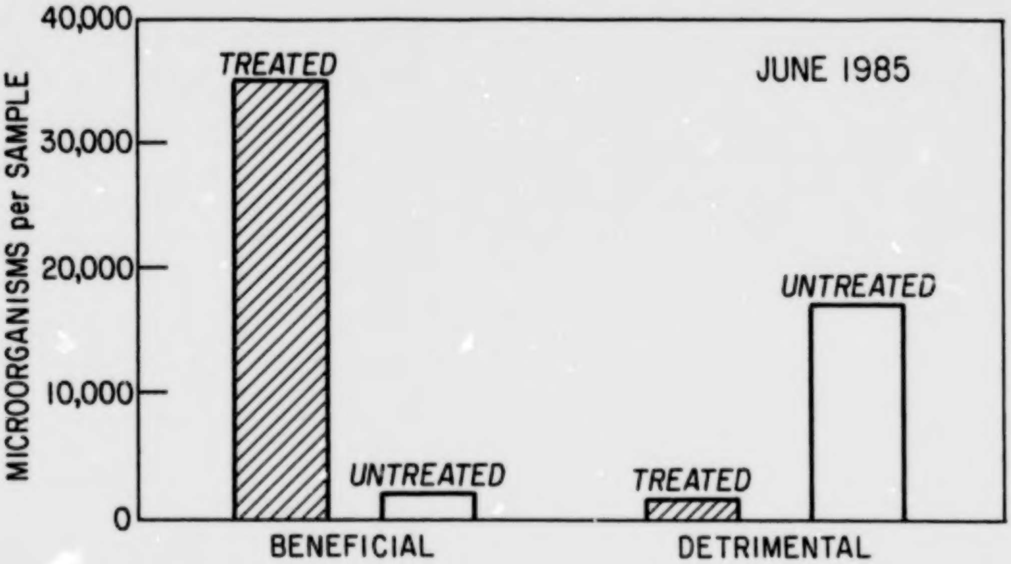


Figure 7. Microbial Activity of Beneficial Heterotrophs and Detrimental T. ferrooxidans 12 Months after Reclamation of Site

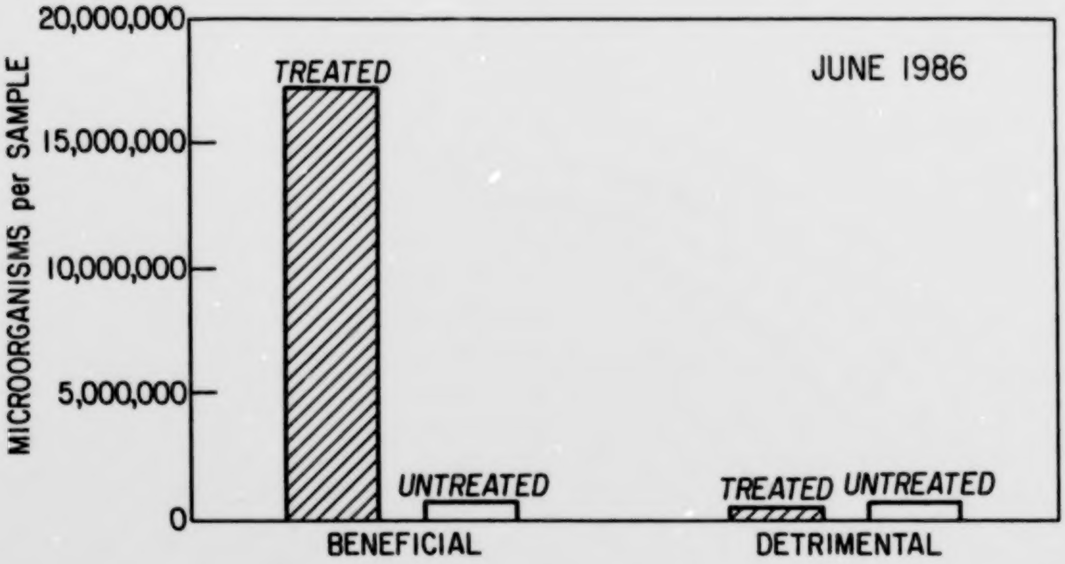


Figure 8. Microbial Activity of Beneficial Heterotrophs and Detrimental T. ferrooxidans 24 Months after Reclamation of Site

(detrimental) on a linear scale. The T. ferrooxidans population in the untreated tract was 170 times that in the treated tract and the beneficial heterotroph population was 30 times that on the untreated tract when checked in June 1985. In June 1986, while the T. ferrooxidans remain suppressed, the heterotrophs not only continue to show the large difference but have increased in their population (500 times) in the treated side.

Water Quality

Drainage Ditch, Installed Drain System & Surface Grade Samples. Water samples were collected throughout the project on a monthly basis; however, no water was collected from the drainage ditch after regrading and recontouring was completed. The background water samples were taken at points that collected drainage (surface and underground) from the entire site. The data (Table 2) indicate that a neutral to slightly alkaline water enters the site and flows past the gob pile. By the time the water leaves the site, the flow has slowed and has become very acid.

After site construction, the previous water flow pattern was not reestablished. All the attempts to obtain water quality samples from the reconstructed drainage ditch and the installed drainpipes (Figure 6) were unsuccessful. Even

Table 2. Background Water Quality Analyses

Parameter	Sample Locations							
	1		2		3		4	
	*	**	*	**	*	**	*	**
pH (field)	2.7	-	2.7	-	2.4	-	6.4	-
pH (lab)	2.6	2.7	2.6	2.7	2.4	2.4	7.0	7.3
Acidity (mg/L)	1034	716	1503	1104	1391	941	-	-
Total Iron (mg/L)	-	72	-	60	-	96	-	0
Temperature (°C)	15	22	15	22	13.5	22	11.5	22
Conductivity (umhos/cm)	2650	2150	3750	2950	3500	2850	650	700

*Water samples taken on 4/1/83
**Water samples taken on 5/18/83
(-) indicates no data

after heavy rainfalls, water quality samples to show the effect of the different treatments applied to the site were not obtained because the installed drains did not work.

Small water samples were taken whenever puddles of standing water were found on site by BFGoodrich personnel. These water samples were only analyzed for the presence of ProMac systems active ingredient because sample size was not large enough to permit acidity and metals analyses. No traces of the active ingredient were detected.

Pressure-Vacuum Lysimeters. In June of 1984, a soil moisture sampling system (pressure-vacuum lysimeters) was installed for monitoring the water quality of the vadose zone nearest the plant root system and as a "back-up" system for the water quality monitoring of run-off and seepage water collected by the drainage ditch. The installation and sampling procedures of Parizek and Lane (1970) were followed.

The data summarized in Tables 3 and 4 are the average of all water quality measurements from samples taken out of the four lysimeters in the treated portion of the site and compared with the average of all water quality measurements from samples taken out of the four lysimeters in the non-ProMac treated portion of the site for year one and year two after reclamation. The data in Table 3 indicate the ProMac treatment has been very successful in reducing all parameters in solution, especially aluminum and manganese for the first year. In the second year (Table 4), the reductions are still large but not as large as in year one except for total iron. Total iron (Table 4) shows a 21% increase in reduction when compared to the iron data in Table 3. Both sets of data show ProMac Systems are effective in reducing acid generation by eliminating T. ferrooxidans.

Figures 9-13 show the differences in the individual parameters (acidity, total iron, sulfates, manganese, and aluminum) when the treated area is compared to the control for the entire length of the project. The treated area parameters show consistently lower concentrations than those in the control area. Only total iron measurements for 12, 15 and 32 months after reclamation are exceptions.

Table 3. Lysimeter Data* Showing Percent Reduction in Treated Area Parameters when Compared with Control One Year after Reclamation

Parameter	Treated Area	Control	Reduction
Acidity (mg/L $CaCO_3$)	1003	3563	72%
Total Iron (mg/L)	236	490	52%
Aluminum (mg/L)	15.3	273.1	94%
Manganese (mg/L)	2.9	43.9	93%
Sulfate (mg/L)	1603	5088	68%

*Data was the average of all measurements from all lysimeters in the treated and the untreated areas for the entire year

Table 4. Lysimeter Data* Showing Percent Reduction in Treated Area Parameters when Compared with Control Two Years after Reclamation

Parameter	Treated Area	Control	Reduction
Acidity (mg/L $CaCO_3$)	1489	3564	58%
Total Iron (mg/L)	375.9	1392	73%
Aluminum (mg/L)	54.1	600.6	91%
Manganese (mg/L)	7.9	74.5	89%
Sulfate (mg/L)	1975	4672	58%

*Data was the average of all measurements from all lysimeters in the treated and the untreated areas for the entire year.

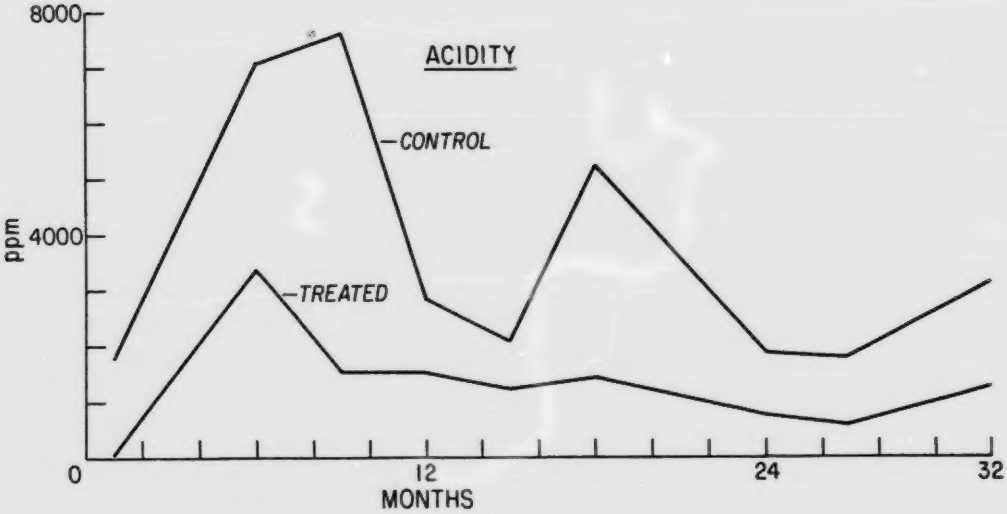


Figure 9. Coal Refuse Acidity Measurement from Lysimeter Water Samples over a 24-Month Period after Reclamation

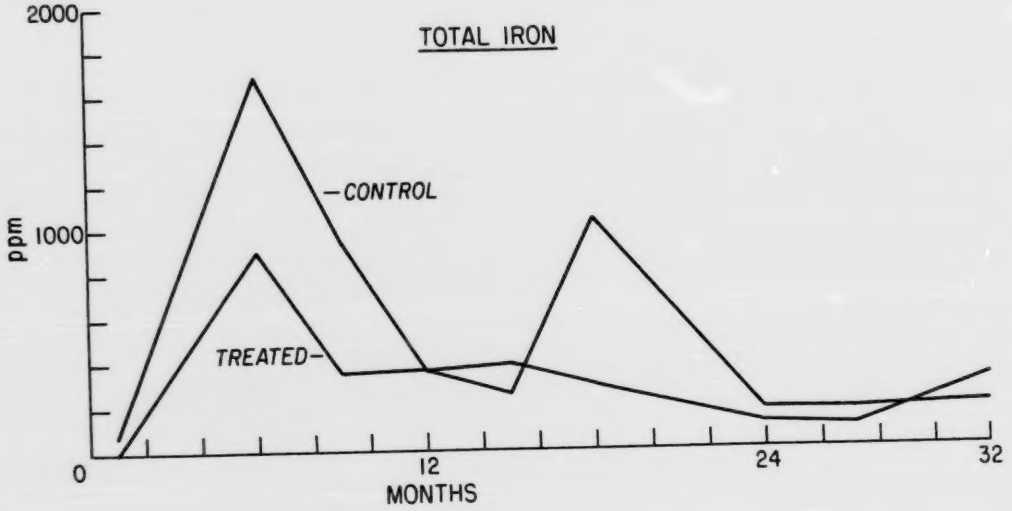


Figure 10. Coal Refuse Total Iron Measurement from Lysimeter Water Samples over a 24-Month Period after Reclamation

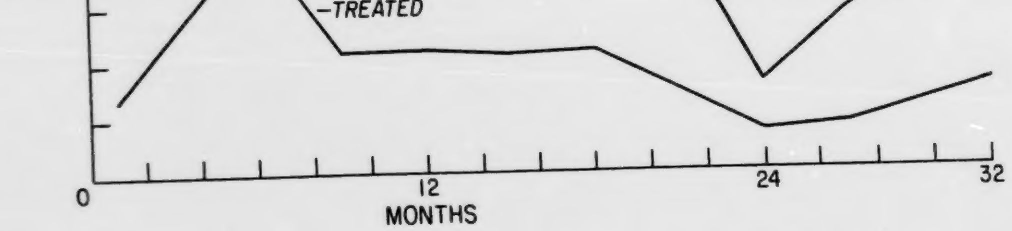


Figure 11. Coal Refuse Sulfates Measurement from Lysimeter Water Samples over a 24-Month Period after Reclamation

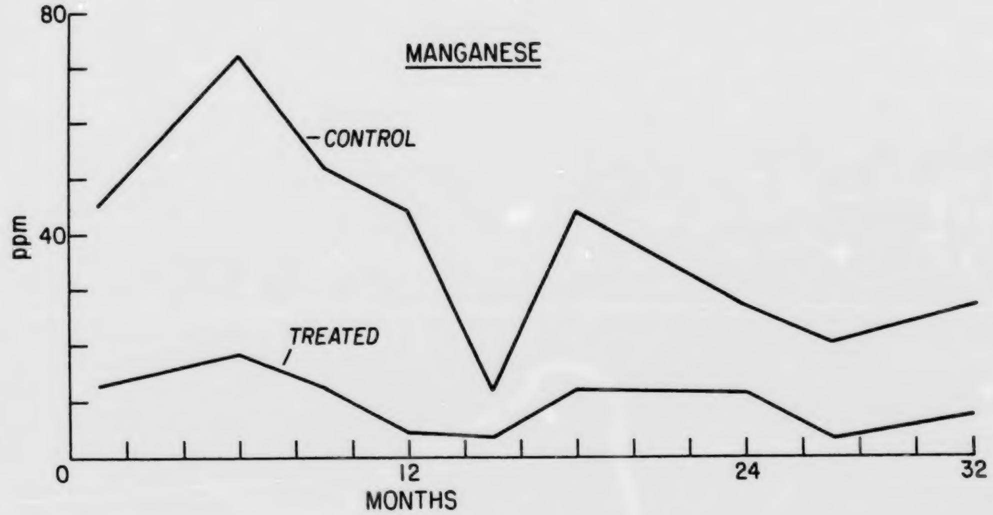


Figure 12. Coal Refuse Manganese Measurement from Lysimeter Water Samples over a 24-Month Period after Reclamation

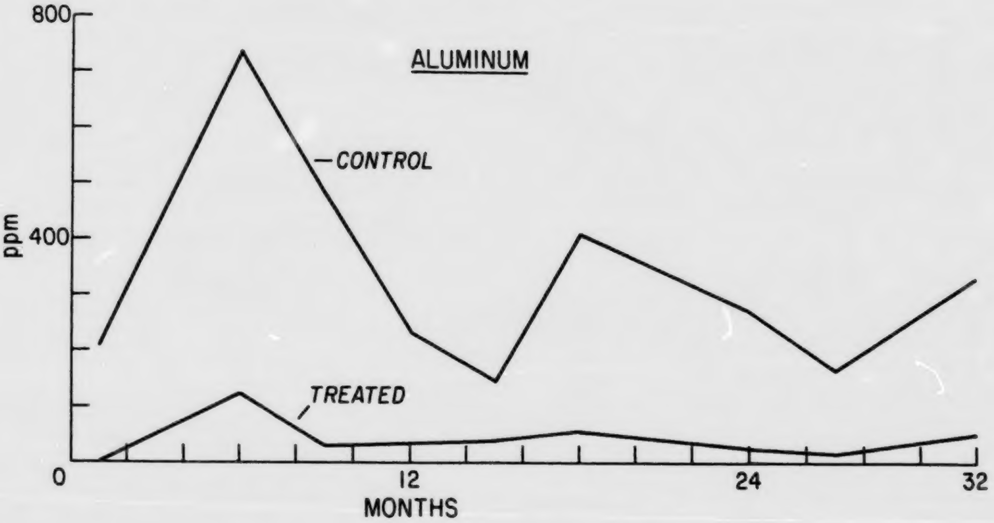


Figure 13. Coal Refuse Aluminum Measurement from Lysimeter Water Samples over a 24-Month Period after Reclamation

Core Samples

Fifty cores were taken at various locations on both the treated and untreated portions of the site at 5 and 24 months after reclamation. Each core sample was measured from the surface of the coal refuse to a depth of 20 cm. Then, each 20 cm core was subdivided into four sections of 5 cm lengths, extracted with water and analyzed. The surface 5 cm of gob should be the area of most activity of T. ferrooxidans; therefore, the comparisons between the 5-month and 24-month data are given in Table 5. The data for the 5-15 cm coal refuse are given in Table 6.

The data in Table 5 indicate a difference between the treated and untreated coal refuse at five months following reclamation; however, after 24 months, the differences between treated and untreated samples were smaller. This trend holds true for the 5-15 cm samples as well (Table 6). There is very little difference in the analysis of the 5 cm material (Table 5) after 5 months or after 24 months.

Summary and Conclusions

The bactericide systems applied to the Route 43 site have substantially reduced acid formation during the project and continue to be effective to the present, 32 months after reclamation. This feat was accomplished without disrupting

Table 5. Core Analyses Data 5 Months and 24 Months after Reclamation at the 0-5 cm Refuse Depth

	Treated		Untreated	
	5 Mo.	24 Mo.	5 Mo.	24 Mo.
Acidity (mg/L $CaCO_3$)	142	110	213	167
pH	3.2	3.4	2.9	3.2
Total Iron (mg/L)	3.9	5.4	17.7	8.1
Aluminum (mg/L)	5.0	2.5	9.7	6.8
Manganese (mg/L)	2.0	1.4	2.5	2.5

Table 6. Core Analyses Data 5 Months and 24 Months after Reclamation at the 5-15 cm Refuse Depth

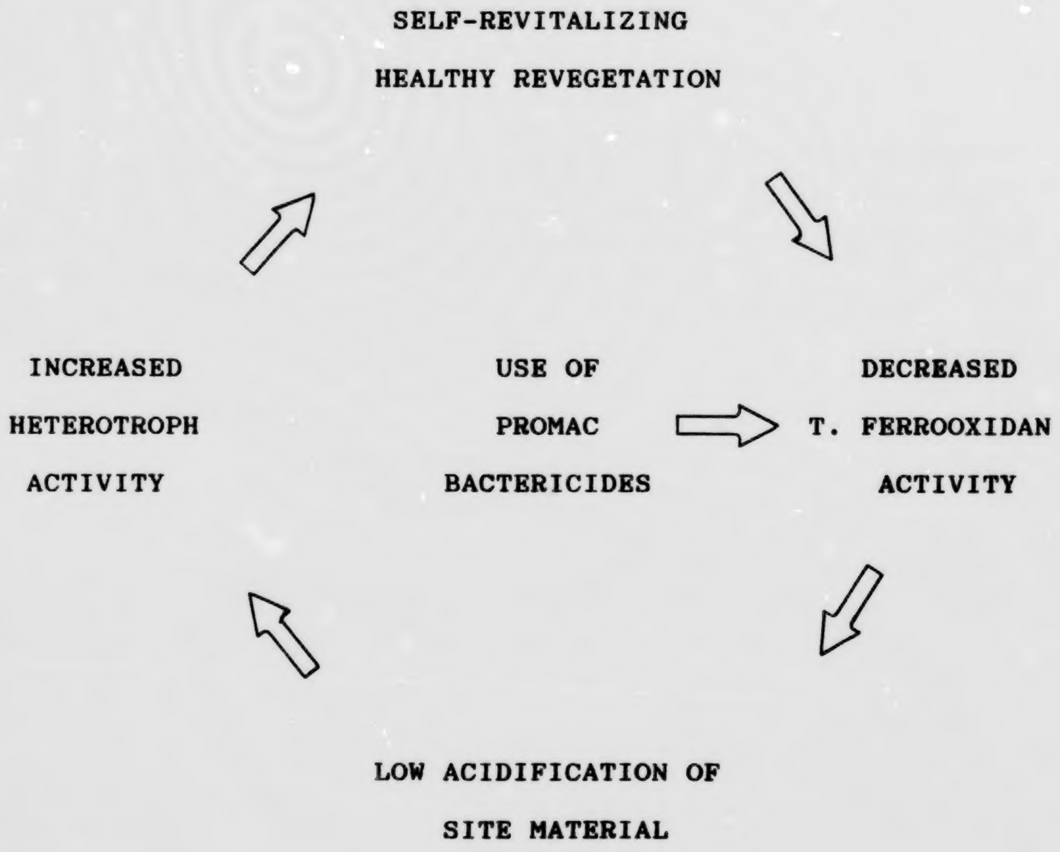
	Treated		Untreated	
	5 Mo.	24 Mo.	5 Mo.	24 Mo.
Acidity (mg/L $CaCO_3$)	140	114	171	155
pH	3.2	3.3	3.1	3.2
Total Iron (mg/L)	5.6	5.3	16.9	7.6
Aluminum (mg/L)	3.5	2.2	5.3	5.4
Manganese (mg/L)	0.6	1.0	1.5	1.9

normal reclamation practices and without ProMac Systems creating a water pollution problem. Water quality samples obtained from the vadose zone using pressure-vacuum lysimeters showed considerable reductions in the parameters measured for the treated area when compared to the untreated control over the entire 24 months of the project. Soil chemical analyses and microbiologic studies of the site prove that ProMac systems eliminate *T. ferrooxidans* and inhibit acid production from the oxidation of pyritic materials which solubilize metal ions such as Fe, Al and Mn.

The beneficial effects of using an anionic surfactant-bactericide for reclamation of mining wastes are the elimination of *T. ferrooxidans*, inhibiting acid formation, and the growth of heterotrophic microorganisms that aid plant growth. Costs are lowered because less non-toxic "topsoiling" materials are needed to ensure successful revegetation. The area treated with the ProMac systems had an average of 3.5 cm less "topsoil" cover than the untreated area. This area has a stable, healthy vegetation and the untreated area does not.

The 24-month bacteria population data show that *T. ferrooxidans* population remained depressed; however, there is no longer a difference between the ProMac systems area and the control area as existed 12 months earlier. The damage done to the control area during the previous 23 months proves that the ProMac systems have protected the area where they were applied and are the only reason for the differences in the vegetation on the Route 43 site.

The beneficial heterotrophic microorganism populations have continued to grow during the 24 months of the project. There was a 500-fold increase in population density from June 1985 to June 1986. Thus, the ProMac systems application has started the site back on the road to full recovery, a part of nature's biological cycle:


SELF-REVITALIZING
HEALTHY REVEGETATION
INCREASED
HETEROTROPH
ACTIVITY
USE OF
PROMAC
BACTERICIDES
DECREASED
T. FERROOXIDAN
ACTIVITY
LOW ACIDIFICATION OF
SITE MATERIAL

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PREVENTION OF ACID GENERATION IN BASE METAL TAILINGS AND WASTE ROCK

A State-Of-The-Art Review for the
Acid Mine Drainage Seminar
Halifax, Nova Scotia
March 1987

by:

Alan V. Bell, P.Eng.
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ABSTRACT

The Canadian metal mining industry has become reasonably proficient over the past decade at the control and treatment of acid drainage resulting from sulphide tailings and waste rock disposal. It has only recently begun to address the issue of preventing acid generation in sulphide wastes in a concerted manner.

The paper reviews the options for the abatement of acid generation and applies these to the commonly available disposal methods for sulphide tailings and waste rock. Preventative techniques based on the current state-of-the-art are recommended.

1.0 INTRODUCTION

At the current time, we have only very limited options for preventing the generation of acid in sulphide bearing tailings and waste rock.

We do have the means to control the impact of acid drainage from tailings and waste rock disposal areas but control is very different from prevention.

Industry and government have recognized in the past 2 to 3 years that finding the means to prevent acid generation in tailings and waste rock will require a concerted national effort. In 1985, the Reactive Acid Tailings (RATS) Program was launched with a view to increasing our understanding of acid generation in tailings and to providing improved predictive and preventative techniques. This program will be described in a separate presentation by Roy John of CANMET.

I am optimistic that the RATS Program will prove to be of great benefit but would like to interject the opinion that we are unlikely to ever see a single solution to acid waste management. It is far more likely that we will develop a number of preventative and control techniques applicable to a wide range of specific situations. These techniques, coupled with improved predictive capabilities, should enable us to address the management of acid wastes far better than we do today.

2.0 HISTORICAL PERSPECTIVE - ACID DRAINAGE IN CANADA

It is useful to briefly look at some key milestones in our recognition of the nature and implications of acid drainage at metal mines in Canada in order to appreciate that we are only now just beginning to look at preventative techniques for both tailings and waste rock:

1969	Heath Steele Mines, New Brunswick: Design of the first comprehensive acid drainage control system in Canada in response to salmon avoidance reactions in the NW Miramichi River.
1970-72	Northeastern New Brunswick Minewater Quality Program: A federal-provincial study resulting in the identification of acid drainage control and treatment measures required to protect salmon streams.
1973	Mine Wastewater Pilot Treatment Project: A 4 gpm treatment plant established at Brunswick Mining with federal funding assistance to establish best practical technology for mine waste waters. The Metal Mining Liquid Effluent Guidelines were derived from this project.
1974-80	Construction of numerous mechanical plants to treat acid drainage at mining operations in Canada.
1983	Recognition of the long-term implications of controlling acidic drainages as an industry/national priority.
1984	Completion of national Sulphide Tailings Management Study.
1985	Establishment of Reactive Acid Tailings (RATS) Program.
1987	Inclusion of acid waste rock into RATS Program.
1990	Scheduled completion of RATS Program.

3.0 ACID TAILINGS & WASTE ROCK CHARACTERISTICS

3.1 Location

The Sulphide Tailings Management Study completed in June 1984 identified a total area of about 10,000 hectares in which sulphide tailings have been, or are, deposited in Canada. Sites were identified in New Brunswick, Quebec, Ontario, Manitoba, British Columbia, the

Yukon, and Northwest Territories. A number of areas in which sulphide tailings had been deposited either in lakes or coastal waters were also identified.

The status of acid generating waste rock disposal in Canada will be known only after the currently ongoing state-of-the-art review, being undertaken for CANMET by Nolan, Davis & Associates, is completed in June 1987. However, it can be anticipated that acid waste rock will be associated with many of the same properties identified in the tailings study.

3.2 Chemical and Physical Properties

Sulphide tailings consist of finely ground solids containing host rock, ferrous sulphides (pyrite, arsenopyrite, pyrrhotite, for example) and at least traces of other base metal sulphides. In some cases, the tailings are readily dewatered by natural draining, whereas in others the grind is very fine, resulting in slimes which tend to retain moisture and therefore do not dewater upon standing. The tailings may contain from only a few percent to over 70 percent of pyrite and other sulphides.

The characteristics of tailings at a given mine may vary over time depending on the uniformity of the ore body, changing beneficiation methods, market conditions, and waste management considerations.

The most apparent difference between sulphide tailings and waste rock is the physical particle size which, in the case of tailings, seldom exceeds 1 mm and is often as low as minus 0.01 mm. Waste rock, on the other hand, can exceed 1 m in particle dimension and may have a widely varying gradation of sizing depending on blasting, extraction, and transportation techniques as well as the geophysical nature of the material in question. Waste rock in some situations represents non-economic low grade ore. In others, it is mineralogically distinct from the ore and its acid generation characteristics may therefore be quite different. It is not uncommon for there to be several distinct types of waste rock at a given property, each with different acid generation characteristics. An excellent example of this is the Equity Silver property in B.C. (Patterson, 1986).

4.0 ABATEMENT OF ACID FORMATION

The formation of acid within sulphide materials is a complex function of many variables ranging from the mineralogy of the deposit to the extent to which organic substances are released by bacteria not directly involved in the oxidation processes (Wakao, 1983; Groudev, 1983). Known factors which influence acid formation include:

- morphology of the sulphide minerals
- particle size of the minerals

- salt and nutrient content
- temperature
- pH
- oxidation/reduction potential
- humidity within the reaction zone
- rate of transport of oxygen to the reaction zone
- rate at which reaction products are transported away from the reaction zone
- catalytic materials (bacterial and other).

The inherent nature of tailings and waste rock deposits (large volumes and surface areas) often enhances conditions favourable to acid formation. Thus, it has been difficult to develop and apply practical abatement methods despite considerable knowledge of the key factors involved in acid generation.

4.1 Limiting the Essential Reactants

Air (oxygen) and moisture are essential reactants for the oxidation of sulphide materials. Elimination of either or both of these reactants prevents the generation of acid.

Of these two essential reactants, complete elimination of water is normally regarded as impractical due to the very small quantity of water required for the oxidation reactions. For example, to generate an acidity of 5000 mg/L (as $CaCO_3$) by oxidation of pyrite, approximately

10 L of air are required, but only 1.6 ml of water enters the reaction. Water is normally diverted away from tailings and waste rock areas more as a means of minimizing the transport of contaminants from the oxidation area, rather than as a means of inhibiting the oxidation reaction (Monenco, 1984).

Observations of the processes occurring within existing tailings deposits demonstrate the importance of limiting oxygen as a reactant. At Waite Amulet (Dave et al., 1986) and Heath Steele (Boorman & Watson, 1974), for example, it has been shown that oxidation occurs only within the uppermost layer of tailings. This is directly attributable to the lack of oxygen at the lower depths. Similarly, observations of tailings that have been disposed of under flooded conditions indicate that essentially no oxidation has occurred. This can be attributed to the limited concentration of oxygen in water and the very low transport rate of oxygen through a layer of water to the reaction zone (Ohio State, 1971).

4.2 Inhibitors

An alternative approach to limiting the essential reactants is the use of methods which will inhibit the sulphide oxidation reaction and thus reduce (or ideally eliminate) the formation of acid.

The use of anionic surfactants, and in particular sodium lauryl sulphate (SLS) has been investigated by the US Bureau of Mines as a bactericide (Kleinmann et al., 1982; 1981). This and other anionic surfactants are bactericidal towards Thiobacillus ferrooxidans and have been shown to reduce acid formation following their application. This and other substances may be useful as inhibitors while revegetation or other more long-term stabilization methods are applied to sulphide wastes.

Other chemicals, such as phosphates, silicates (Watkin and Watkin, 1983) and sodium chloride have also been proposed as inhibitors. Phosphates have been shown to reduce the rate of acid generation. This is attributed to the precipitation of iron phosphates which remove iron as a potential oxidant of pyrite. Other agents, such as silicates and iron hydroxide precipitates, may act as inhibitors by coating the sulphide particles, thereby limiting the access of air. At present, these reagents have been proposed primarily as temporary inhibitors of acid formation to be used as an aid to revegetation efforts. The use of sodium-chloride-rich clays has been proposed as a sealant and bacterial inhibitor for pipeline excavations (Nolan, Davis & Associates, 1984). As reported elsewhere at this seminar, sodium chloride has also been applied in association with a clay cover on a pyritic slate waste dump at the Halifax International Airport.

The creation of anaerobic conditions in sulphide wastes by the inclusion of organic material is another option which may have application in some circumstances and, therefore, warrants further evaluation.

4.3 Revegetation

Any discussion on the abatement of acid drainage from mine wastes would be seriously lacking if it failed to address the issue of revegetation. The emphasis in the management of sulphide minewastes to date has been directed towards the establishment of vegetative covers on tailings and waste rock disposal areas with a view to meeting the following objectives:

- stabilizing surface materials to wind and water erosion
- maximizing evapotranspiration to reduce the quantity of infiltration and runoff
- reducing oxygen transfer into the waste
- improving the aesthetic value of the site
- improving biomass productivity of the site.

Despite the very hostile conditions that most tailings and waste rock disposal areas present to the establishment of vegetation, it is fair to say that the mining industry has developed a wide range of site-specific techniques which have enabled vegetation to be established in most situations. The cost of this achievement and, in some cases, of sustaining plant growth, is often very high. The effect that vegetative covers have on impeding the oxidation of underlying wastes is currently being addressed as a function of the RATS Program. Without wishing to prejudge this work, there is little evidence at this time that vegetation in itself has a significant effect on reducing oxygen transfer although it certainly meets a number of the other objectives described above.

5.0 DISPOSAL ALTERNATIVES

The alternative methods of tailings disposal used at base metal properties in Canada include the following:

1. Discharge as a slurry into a specifically designed containment area. Subcategories include:
    a) discharge into a tailings pond (subaqueous discharge)
    b) subaerial discharge methods onto tailings beaches including point discharge, spigotted, cycloned, or layered tailings deposition.
    c) disposal as thickened tailings in free-standing mounded deposit (Robinsky method).
2. Underwater disposal in lakes or coastal waters.
3. Use as mine backfill.

The implications of each of these methods from the operational and environmental standpoint are reviewed in the recent update of the EPS Mine and Mill Wastewater Treatment Manual (Monenco, 1986).

Waste rock disposal alternatives include:

1. Land-based disposal in engineered dumps.
2. Backfill into open-pit or underground mines.
3. Underwater disposal where applicable.

Given these basic disposal alternatives for tailings and waste rock disposal, what are the options for minimizing or ideally, preventing the generation of acid?

Tailings Disposal:

At the current time, the only practical and proven long-term approach to controlling the formation of acid in sulphide tailings is to limit the availability of oxygen as a reactant by maintaining the waste in a saturated or submerged condition.

Thus, the options available include either disposal into lakes or coastal waters, or the use of engineered containment areas in which the tailings are discharged subaqueously and the system designed to allow for maintaining the tailings in a saturated condition in perpetuity. Depending on the local hydrologic regime, this could represent a maintenance-free alternative in many circumstances. Site selection, understanding the local hydrologic and hydrogeological characteristics and design of the retaining structures are extremely important parameters in meeting this objective.

It is possible that research being undertaken as a function of the RATS Program will determine other means of limiting availability of oxygen to tailings deposits but at this time, water is the only practical alternative that is generally available. In arid areas, it may be possible to seal tailings deposits sufficiently to limit the availability of water for the transport of the products of oxidation within tailings deposits, and thus effectively control the impact of acid generation. Similarly, in northern areas, it may be feasible to encourage the formation of permafrost in tailings deposits and thus effectively seal the system. Application of these two approaches, if practical, is obviously geographically limited.

Methods of tailings disposal which allow for large areas of tailings to be exposed to oxidation on beaches with periodic flushing by precipitation and runoff should normally be avoided from the standpoint of minimizing acid generation. It may be possible to control the effects of acid generation during the operational phase but under most circumstances, these subaerial disposal options will result in a final configuration from which it will be very difficult to seal both air and water penetration.

Layered deposition of tailings as practiced at the Westmin Mine in B.C. and elsewhere produces a very dense tailings deposit of low permeability which may prove beneficial to the long-term management of acid tailings. However, this approach poses operational difficulties under winter conditions and is, therefore, likely to be limited to more temperate regions.

Waste Rock Disposal:

Although the biochemistry of acid generation in sulphide waste rock may be very similar to that in tailings, the physical differences, more limited range of disposal options, and the fact that far less work has been done on long-term management, result in waste rock posing an even greater challenge in the search for preventative techniques.

The cost of moving waste rock has a major influence on the economics of most open pit operations and disposal sites are always sought with a view to minimizing haul distances and grades. Double handling of waste rock is seldom practical or desirable unless there is an opportunity to use the material for backfill in underground mining situations. Where sulphide bearing material is used as backfill, precautions must be taken to ensure that conditions conducive to exothermic oxidation are avoided.

The location of some properties may provide opportunities for the disposal of waste rock under water, either through dumping into lakes or coastal waters, or into worked out pits which can subsequently be flooded. Although the subaqueous disposal of sulphide bearing waste rock is attractive from the standpoint of minimizing acid generation and its impact, such disposal options will often cause concern over other potential impacts such as sedimentation or habitat destruction. The Cape Breton Development Corporation has experienced such difficulties relative to its use of marine disposal of waste rock, for example.

In the majority of cases, the only practical alternative for waste rock disposal is an engineered waste dump. These often take the form of side-hill systems although four-sided waste piles on relatively flat terrain are not uncommon. The Acid Waste Rock Management Study currently in progress (Nolan, Davis & Associates, 1987) will provide more complete inventory data than is presently available; but to the knowledge of the writer, there are no sulphide waste rock disposal systems in Canada or elsewhere where the system has been designed to successfully prevent the generation of acid.

In addition to the net acid generating potential of the waste rock to be stored, the key characteristic influencing the generation and control of acid is the porosity of the waste in place. Dumps with high void ratios lead to minimal groundwater mounding within the dump but can develop potentially serious problems with indrawn air due to higher than ambient temperatures being created in the dump by the oxidation reaction. This latter situation exists at the Equity Silver Dump in B.C. where attempts to prevent indrawing of air have been made by placing a continuous clay seal on the side slopes of the dump. Harries and Richie (1986) report a controlled experiment of the effect of sealing two pyritic waste dumps in Australia and showed that reduced oxygen levels in the pore spaces of the dump led to a reduction in the oxidation rate.

Oxidation inhibitors have not been shown to have long-term benefits in preventing acid generation in waste dumps although their use to control oxidation rates while more permanent measures are implemented has proven to be valuable at a number of locations.

At the present state-of-knowledge, it appears that the best generic approach to minimizing the generation of acid in sulphide waste dumps is to construct the dump on a cellular basis incorporating the following features:

- placement of the waste in uniform layers with compaction to reduce the pore space and thus minimize air convection and oxygen transfer within the pile.
- placement of continuous horizontal and vertical barriers of compacted impermeable till to reduce the ingress and movement of moisture and air within the pile.
- use of bactericides or other inhibitors on the waste prior to placing the next impermeable layer.
- sealing and vegetation of side slopes as soon as they become inactive.
- final capping and doming of the waste pile with impermeable material.
- establishment of self-sustaining vegetation on the final cover.

This is shown conceptually in Figure 1.

The practicality of this approach will, of course, depend on the availability of impermeable materials within a reasonable distance of the site.

It must be emphasized that there is no evidence to date that such measures will absolutely prevent acid generation. Thus, the siting and design of the dump so as to facilitate the collection and control of any contaminated surface and groundwaters are of paramount importance.

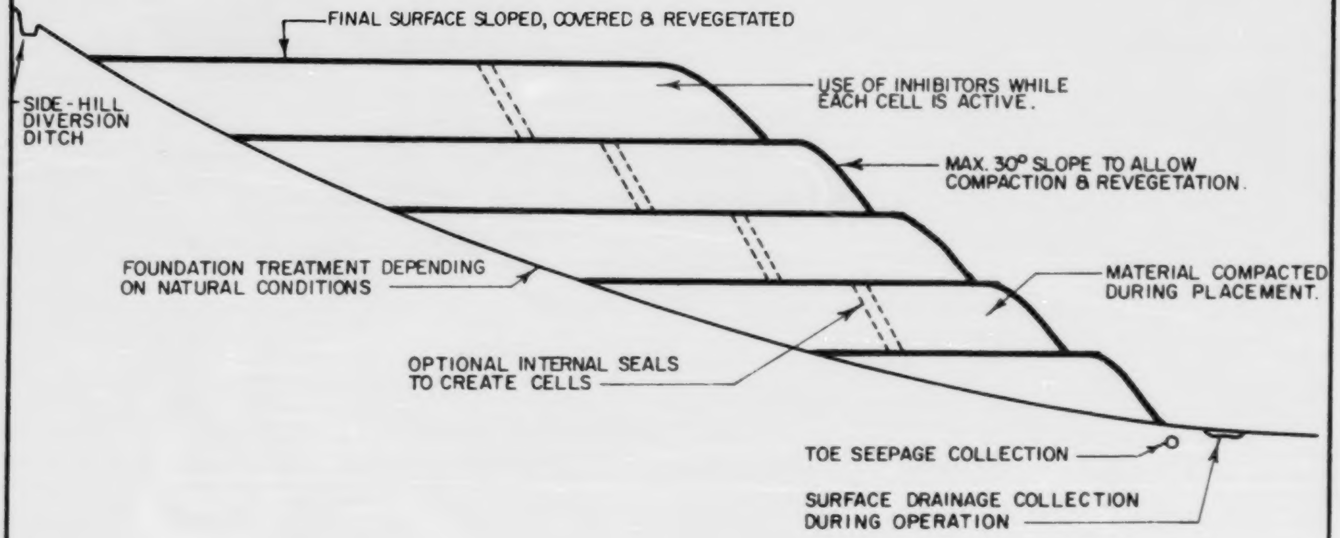
In this regard, it becomes particularly important that we improve our ability to predict the way in which a given type of sulphide-bearing waste rock is likely to become acid producing. Only in this way will we be able to truly evaluate factors such as the long-term economic implications of waste rock disposal to mine development, and to compare and select different disposal options. The same can be said for sulphide tailings, although more work has been done on predictive techniques for tailings than for waste rock.

6.0 CONCLUSIONS

1. We have become reasonably proficient over the past decade at the control and treatment of acid drainage but are only just beginning to address the issue of prevention at a practical level relative to sulphide tailings and waste rock.

2. Utilization of disposal methods for sulphide tailings which maintain the material in a submerged or saturated condition is the best long-term preventative technique currently available.

3. Use of engineered disposal areas, or dumps, is likely to be the only practical alternative for waste rock at most mine sites. These systems can be designed to minimize acid generation but as yet there is no evidence that the need to control contaminants from these systems can be entirely avoided in most circumstances.


FINAL SURFACE SLOPED, COVERED & REVEGETATED
USE OF INHIBITORS WHILE EACH CELL IS ACTIVE.
SIDE-HILL DIVERSION DITCH
MAX. 30° SLOPE TO ALLOW COMPACTION & REVEGETATION.
FOUNDATION TREATMENT DEPENDING ON NATURAL CONDITIONS
MATERIAL COMPACTED DURING PLACEMENT.
OPTIONAL INTERNAL SEALS TO CREATE CELLS
TOE SEEPAGE COLLECTION
SURFACE DRAINAGE COLLECTION DURING OPERATION

NOLAN, DAVIS
& ASSOCIATES

FIGURE I

ENGINEERED SIDE-HILL DUMP

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AQUATIC PLANTS AND BOG COVERS TO PREVENT ACID GENERATION - BASE METAL TAILINGS

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ABSTRACT

The reactions involved in bacterial metal sulfide mineral oxidation and the factors affecting them are described in relation to acid generation in base metal tailings. The results of laboratory leaching tests on tailings innoculated with iron oxidizing bacteria indicate that placement of oxygen-scavenging material on the surface of these tailings arrest bacterial sulfide mineral oxidation and thus eliminate sulfuric acid generation.

INTRODUCTION

Bacterial sulfide mineral oxidation is the principal cause of sulfuric acid generation in base metal tailings. The requirements for sulfide mineral oxidation is the same as for any other type of oxidation, including fire: an oxidizable substrate or fuel; an oxidant; and a physical or chemical promotor (Figure 1). In the case of fire, heat acts as the promotor; in biological oxidations, either enzymes or microorganism serve this function. The chemical reactions of bacterially mediated mineral oxidation and the factors that influence them will be described and a method in which these reactions are retarded in base metal tailings by the placement of

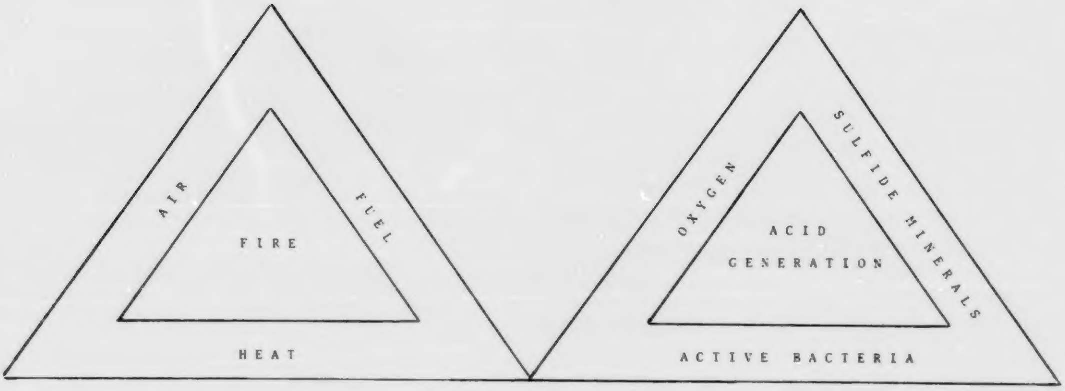

Figure 1. Comparison of requirements for fire and bacterial metal sulfide mineral oxidation.

vegetated wetland soil on the surface of these tailings will be demonstrated.

Reviews on the interaction of iron oxidizing bacteria have recently been written (Harrison, 1984; Hutchins *et al*, 1986; Karavaiko and Torma, 1985; Lundgren and Silver, 1980; Torma and Bosecker, 1982). Reference should also be made to a good textbook on hydrogeochemistry (Drever, 1982), the ecology of wetlands (Mitsch and Gosselink, 1986) and a number of symposia on the interaction of wetlands with mining activity (Brooks *et al*, 1985; Graves 1983; 1984; 1985).

BACTERIA

The most important bacteria in the oxidation of sulfide minerals are the iron oxidizing thiobacilli commonly called by the binomial *Thiobacillus ferrooxidans*. These bacteria are obligate autotrophs; they require an inorganic source of energy - either ferrous iron or reduced sulfur compounds - and an inorganic source of carbon - CO_2. Most organic compounds are toxic and growth is stimulated in the presence of bacteria which can remove these inhibitors. These bacteria are obligate aerobes; oxygen is required not only as a terminal acceptor of electrons from ferrous iron or sulfite, but also for the oxygenation of elemental sulfur and sulfide. They are also obligate acidophiles, their growth being limited to pH values of 1 - 6.5. Although the iron oxidizing thiobacilli have been shown to fix molecular nitrogen, optimal growth is obtained in the presence of ammonia or urea. Harrison (1984) has confirmed the hypothesis originally foreworded by Margalith _et al_. (1966) that the iron oxidizing thiobacilli are a group of nutritionally similar species. The iron oxidizing thiobacilli have a very high tolerance to heavy metal inhibition which may be plasmid encoded.

Morphologically and physiologically similar to the iron oxidizing thiobacilli is _Thiobacillus thiooxidans_. _T. thiooxidans_ is not related to the iron oxidizing thiobacilli as the guanosine and cytosine content of its DNA is 50-51% compared to 55-60% for the iron oxidizers. _T. thiooxidans_ is more acid tolerant than the iron oxidizing thiobacilli, and is more active in oxidizing elemental sulfur, sulfide and partially oxidized oxyions of sulfur; it is an obligate aerobic autotroph and is inhibited by most organic compounds. It is found in association with the iron oxidizing bacteria and stimulates the oxidation of pyrite and chalcopyrite. Other thiobacilli found in association with the iron oxidizing bacteria are the acidiophillic facultative autotrophs _Thiobacillus acidophilus_ and _Thiobacillus organoparus_, the obligatly autotrophic neutrophillic Thiobacillus thioparus and Thiobacillus neapolitanus, and the facultative anaerobe Thiobacillus dentrificans.

Of uncertain importance in the oxidation of sulfide minerals in nature is the iron oxidizing bacteria Leptospirillum ferrooxidans, which can oxidize iron but not sulfur compounds. Facultative thermophillic iron oxidizing bacilli and lobate iron oxidizing bacteria have also been isolated.

CHEMICAL AND BIOLOGICAL REACTIONS

The oxidation of sulfide minerals by bacteria has been attributed to two mechanisms which may not be mutually exclusive. In the direct mechanism, the iron moiety, the sulfide moiety, or both if they occur in the same mineral, are oxidized directly by the bacteria with the formation of the metal sulfate.

$$MS + O_2 \rightarrow MSO_4$$

The indirect mechanism consists of the oxidation of the minerals by soluble ferric iron, present as ferric salts, usually as the sulfate; the products of this oxidation are the sulfates of metals in the mineral, ferrous sulfate and elemental sulfur. The following equation illustrates this mechanism:

$$MS + Fe_2(SO_4)_3 \rightarrow MSO_4 + FeSO_4 + S^o$$

The involvement of the bacteria would be to reoxidize the ferrous iron

$$4\ FeSO_4 + O_2 + 2\ H_2SO_4 \rightarrow 2\ Fe_2(SO_4)_3 + 2\ H_2O$$

and the elemental sulfur

$$2\ S^o + 3\ O_2 + 2\ H_2O \rightarrow 2\ H_2SO_4$$

Because of the extremely low solubility of the heavy metal sulfides, intimate contact was suggested to be required for the bacteria to oxidize these minerals. Intimate contact of the bacteria has been shown to occur in the presence of such sulfide minerals as pyrite, chalcopyrite and galena and in the presence of elemental sulfur. Torma and Sakaguchi (1978) however, reported that the solubilization of metals from sulfide minerals was dependent upon the solubility constants of these minerals. Although in the past the indirect mechanism was thought to be the principle if not the only method of solubilization from ores, subsequent work on the oxidation of natural and synthetic metal sulfides devoid of iron proved beyond all doubt that direct attack on these minerals by the bacteria could occur. Further evidence for the direct mechanism was demonstrated by leaching studies on chalcopyrite and pyrite using specific metabolic inhibitors. N-ethyl maleimide inhibits the oxidation of the sulfide moiety and sodium azide inhibits the oxidation of the ferrous iron. Various researchers have shown that the iron oxidizing bacteria could oxidize both the iron and sulfur moieties from chalcopyrite and pyrite. Although the rates of these oxidations varied with the treatment of both the bacteria and minerals, the rates of oxidation of the iron and sulfide moieties were found to be additive.

In contrast to the direct mechanism, the indirect mechanism of bacterial metal solubilization is not limited to metal sulfides; arsenides, carbonates, oxides and silicates may be altered to yield soluble metals. These reactions are caused by chemical interactions between the minerals and products of bacterial metabolite activities, the most important of which are ferric iron and sulfuric acid. When ferric iron oxidizes the minerals, ferrous iron and elemental sulfur are formed. The sulfur can be oxidized to sulfuric acid and the ferrous iron can be oxidized to ferric iron at a rate estimated by Lacey and Lawson (1970) at a rate some 500,000 times that of the abiological rate. The iron oxidizing enzyme system is inhibited by high concentrations of the substrate, ferrous iron and by high concentrations of the product, ferric iron. This inhibition is not due to osmotic effects as iron oxidation has been shown to proceed in the presence of salt concentrations as high as one molar. Neither an abundant carbon supply in the form of CO_2 or conditions for rapid growth are necessary for the bacterial oxidation of ferrous iron.

THE OXIDATION OF IRON SULFIDE MINERALS

As pyrite is the most common iron sulfide mineral found in nature, it has been one most extensively studied. Marcasite is a mineral with the same chemical composition but different crystal structure. Direct oxidation of both of these iron disulfide minerals can be represented by the following series of equations:

$$4\ FeS_2 + H_2O + 14\ O_2 \dashrightarrow 4\ FeSO_4 + 4\ H_2SO_4$$

$$4\ FeSO_4 + O_2 + H_2SO_4 \dashrightarrow 2\ Fe_2(SO_4)_3 + 2\ H_2O$$

$$\overline{4\ FeSO_2 + 15\ O_2 + 2\ H_2O \dashrightarrow 2\ Fe_2(SO_4)_3 + 2\ H_2SO_4}$$

The pyrite is first oxidized directly by the bacteria with the formation of ferrous sulfate, which is subsequently oxidized to ferric sulfate. The ferric iron thus formed could also oxidize the pyrite under the proper conditions, that is, at low pH values when the mineral surfaces are unobstructed by chemical precipitates. The oxidation of the pyrite by soluble ferric iron may be represented by the equation:

$$2\ FeS_2 + 2\ Fe_2(SO_4)_3 \rightarrow 6\ FeSO_4 + 4\ S^{o}$$

The resulting ferrous iron could then be reoxidized to ferric iron by the iron oxidizing thiobacilli, and the elemental sulfur to sulfuric acid by either or both of the iron oxidizing thiobacilli and other acidophilic thiobacilli.

Troilite and mackinawite, two iron sulfide minerals containing iron and sulfur in equimolar proportions, can be oxidized directly according to the equation for

$$\begin{array}{c} FeS + 8\ O_2 \rightarrow 4\ FeSO_4 \\ 4\ FeSO_4 + O_2 + 2\ H_2SO_4 \rightarrow 2\ Fe_2(SO_4)_3 + 2\ H_2O \\ \hline 4\ FeS + 9\ O_2 + 2\ H_2SO_4 \rightarrow 2\ Fe_2(SO_4)_3 + 2\ H_2O \end{array}$$

or indirectly according to the equation

$$FeS + Fe_2(SO_4)_3 \rightarrow 3\ FeSO_4 + S^{o}$$

Pyrrhotite, a sulfur-rich iron sulfide mineral is also oxidized directly by the iron oxidizing bacteria according to the equation:

$$4\ Fe_7S_8 + 69\ O_2 + 10\ H_2SO_4 \rightarrow 14\ Fe_2(SO_4)_3 + 10\ H_2O$$

or by ferric sulfate

$$Fe_7S_8 + 7\ Fe_2(SO_4)_3 \rightarrow 21\ FeSO_4 + 8\ S^{o}$$

The mechanism of arsenopyrite oxidation by the iron-oxidizing bacteria may be illustrated by the following reaction:

$$2\ FeAsS + 7O_2 + 2\ H_2O \rightarrow 2\ FeAsO_4 + 2\ H_2SO_4$$

and the oxidation of this mineral by ferric sulfate may be illustrated by the following series of reactions:

$$\begin{array}{c} 4\ FeAsS + 4\ Fe_2(SO_4)_3 + 10\ H_2O + 11\ O_2 \rightarrow 4\ H_3AsO_4 + 12\ FeSO_4 + 4\ H_2SO_4 \\ 4\ H_3AsO_4 + 2\ Fe_2(SO_4)_3 \rightarrow 4\ FeAsO_4 + 6\ H_2SO_4 \\ \hline 4\ FeAsS + 6\ Fe_2(SO_4)_3 + 10\ H_2O + 11\ O_2 \rightarrow 4\ FeAsO_4 + 12\ FeSO_4 + 10\ H_2O \end{array}$$

The concentration of soluble arsenic rarely exceeds 1 g/L during bacterial arsenopyrite oxidation due to the secondary formation of iron arsenate, mineralogically known as scorodite, which is sparingly soluble.

Pentlandite, a nickel and cobalt containing sulfide mineral can also be oxidized directly by the iron oxidizing thiobacilli according to the equation

$$8\ (Ni,Co,Fe)_9S_8 + 14\ O_2 + 26\ H_2SO_4 \rightarrow 36\ (Ni,Co)SO_4 + 18\ Fe_2(SO_4)_3 + 26\ H_2O$$

or indirectly by ferric sulfate

$$(Ni,Co,Fe)_9S_8 + 9\ Fe_2(SO_4)_3 \rightarrow 9\ (Ni,Co,Fe)SO_4 + 18\ FeSO_4 + 8S^o$$

OXIDATION OF COPPER SULFIDE MINERALS

Most, if not all sulfide minerals may be oxidized by the iron oxidizing bacteria either directly or indirectly. These reactions have been intensively reviewed by Dutrizac and MacDonald (1974) Lungren and Silver (1980) and Karavaiko and Torma (1985).

A series of reactions has been proposed for the oxidation of chalcopyrite mediated by the iron oxidizing thiobacilli (Lundgren and Silver, 1980). This series of reactions shows both the direct attack on the chalcopyrite by the bacteria and the oxidation of this mineral by soluble ferric iron. As chalcopyrite contains both sulfide and iron, both mechanisms occur separately and are difficult to separate without the use of metabolic inhibitors specific for iron and sulfide oxidation.

$$\begin{array}{c}
48\ CuFeS_2 + 204\ O_2 + 24\ H_2SO_4 \text{ --> } 48\ CuSO_4 + 24\ Fe_2(SO_4)_3 + 24\ H_2O \\
12\ CuFeS_2 + 24\ Fe_2(SO_4)_3 \text{ --> } 12\ CuSO_4 + 60\ FeSO_4 + 24\ S^o \\
60\ FeSO_4 + 30\ H_2SO^4 + 15\ O_2 \text{ --> } 30\ Fe_2(SO_4)_3 + 30\ H_2O \\
24\ S^o + 24\ H_2O + 36\ O_2 \text{ --> } 24\ H_2SO_4 \\
30\ Fe_2(SO_4)_3 + 120\ H_2O \text{ --> } 20\ H[Fe(SO_4)_2.2\ Fe(OH)_3] + 50\ H_2SO_4 \\
\hline
60\ CuFeS_2 + 25\ SO_2 + 90\ H_2O \text{ --> } 60\ CuSO_4 + 20\ H[Fe(SO_4)_2.2\ Fe(OH)_3] + \\
20\ H_2SO_4
\end{array}$$

Initially the bacteria oxidizes the sulfide moiety of the chalcopyrite, solubilizing the copper as cupric sulfate and the iron as ferric sulfate. The ferric sulfate can then oxidize the mineral with the formation of cupric sulfate, ferrous sulfate and elemental sulfur. Both the ferrous sulfate and the elemental sulfur can then be oxidized by the bacteria to ferric sulfate and sulfuric acid respectively. Resulting ferric iron can then either be used for the chemical oxidation of chalcopyrite, or under suitable conditions be hydrated with the formation of secondary insoluble iron compounds such as jarosite. The series of reactions described are idealized and in reality, the stoichiometry will vary greatly under different physical and chemical conditions.

Other mixed copper and iron sulfide minerals can be oxidized either directly or indirectly according to the following equations:

for bornite

$$4\ Cu_5FeS_4 + 37\ O_2 + 10\ H_2SO_4 \text{ --> } 20\ CuSO_4 + 2\ Fe_2(SO_4)_3 + 10\ H_2O$$

$$Cu_5FeS_4 + 6\ Fe_2(SO_4)_3 \text{ --> } 5\ CuSO_4 + 13\ FeSO_4 + 4\ S^o$$

cubanite

$$2\ CuFe_2S_2 + 13\ O_2 \rightarrow 2\ CuSO_4 + 2\ Fe_2(SO_4)_3 + 2\ H_2O$$

$$CuFe_2S_3 + 3\ Fe_2(SO_4)_3 \rightarrow CuSO_4 + 8\ FeSO_4 + 3\ S^o$$

and tetrihedrite

$$2\ (3\ Cu_2S.Sb_2S_3) + 27\ O_2 \rightarrow 12\ CuSO_4 + 2\ Sb_2O_3$$

Chalcocite, the most abudant copper mineral found in nature after chalcopyrite, is oxidized directly by iron oxidizing thiobacilli with the intermediate formation of digenite and covellite. The following series of equations illustrates not only the oxidation of chalcocite but also the oxidation of these other two minerals.

$$\begin{array}{c}
10\ Cu_2S + 2\ H_2SO_4 + O_2 \rightarrow 2\ Cu_9S_5 + 2\ CuSO_4 + 2\ H_2O \\
2\ Cu_9S_5 + 8\ H_2SO_4 + 4\ O_2 \rightarrow 10\ CuS + 8\ CuSO_4 + 8\ H_2O \\
10\ CuS + 20\ O_2 \rightarrow 10\ CuSO_4 \\
\hline
10\ Cu_2S + 10\ H_2SO_4 + 25\ O_2 \rightarrow 20\ CuSO_4 + 10\ H_2O
\end{array}$$

Ferric iron can oxidize these minerals to $CuSO_4$ also, according to the following reactions:

$$Cu_2S + 2\ Fe_2(SO_4)_3 \rightarrow 2\ CuSO_4 + 4\ FeSO_4 + S^o$$

$$Cu_9S_5 + 9\ Fe_2(SO_4)_3 \rightarrow 9\ CuSO_4 + 18\ FeSO_4 + 5\ S^o$$

$$CuS + Fe_2(SO_4)_3 \rightarrow CuSO_4 + 2\ FeSO_4 + S^o$$

OXIDATION OF OTHER MINERALS

Sulfide minerals of cadmium (greenockite), cobalt (sycoporite), nickel (millerite) and zinc (sphalerite) are oxidized both directly and indirectly with the formation of the soluble metal sulfates. Lead sulfide (galena) is also oxidized with the formation of the insoluble sulfate mineral, anglesite. Stibnite, a sulfide mineral of antimony, is oxidized microbiologically by a complex series of reactions described by Torma and Gabra (1977):

$$2\ Sb_2S_3 + 12\ O_2 \text{-->} 2\ Sb_2(SO_4)_3$$

$$Sb_2(SO_4)_3 + 2\ H_2O \text{-->} (SbO)_2SO_4 + 2\ H_2SO_4$$

$$Sb_2(SO_4)_3 + O_2 + H_2SO_4 \text{-->} Sb_2(SO_4)_5 + 2\ H_2O$$

$$Sb_2(SO_4)_5 + 4\ H_2O \text{-->} (SbO_2)_2SO_4 + 4\ H_2SO_4$$

$$2\ Sb_2S_3 + 13\ O_2 + 4\ H_2O \text{-->} (SbO)_2SO_4 + (SbO_2)_2SO_4 + 4\ H_2SO_4$$

Stibnite can also be oxidized by ferric iron according to the equation:

$$Sb_2S_3 + 3\ Fe_2(SO_4)_3 \text{-->} Sb_2(SO_4)_3 + 6\ FeSO_4 + 3\ S^o$$

The iron oxidizing bacteria may directly oxidize uranium minerals such as uraninite (pitchblend)

$$2\ UO_2 + O_2 + 2\ H_2SO_4 \text{-->} 2\ UO_2SO_4 + 2\ H_2O$$

and brannerite

$$2\ UTiO_2 + 3\ O_2 + 2\ H_2SO_4 \text{-->} 2\ UO_2SO_4 + 2\ TiO_2 + 2\ H_2O$$

These minerals may also be oxidized by ferric iron according to the following reactions

$$UO_2 + Fe_2(SO_4)_3 \text{-->} UO_2SO_4 + 2\ FeSO_4$$

$$UTiO_2 + Fe_2(SO_4)_3 \text{-->} UO_2SO_4 + TiO_2 + 2\ FeSO_4$$

Similarly, carbonate minerals can be oxidized by the action of the thiobacilli. This can be illustrated by the oxidation of the iron carbonate mineral, siderite, by the iron oxidizing bacteria

$$4\ FeCO_3 + O_2 + 6\ H_2O \text{-->} 4\ Fe(OH)_3 + 4\ CO_2$$

Sulfur oxidizing bacterial can mediate the solubilization of zinc from the carbonate mineral, smithsonite, according to the equation

$$2\ ZnCO_3 + 3\ O_2 + 2H_2O + 2\ S^o \text{-->} 2\ ZnSO_4 + 2\ H_2CO_3$$

ANEROBIC REACTIONS OF IRON AND SULFUR

Although experimental evidence is still sparse at this time, the possibility exists that sulfur can be oxidized to sulfuric acid under anerobic conditions. Sugio *et al* (1985) have proposed a two step anerobic mechanism which may be present in thiobacilli, including the iron-oxidizing bacteria, for the oxidation of elemental sulfur by ferric iron illustrated in the following series of reactions:

$$\begin{array}{c} S^o + 4\ FeCl_3 + 3\ H_2O \longrightarrow H_2SO_3 + 4\ FeCl_2 + 4\ HCl \\ H_2SO_3 + 2\ FeCl_2 + H_2O \longrightarrow H_2SO_4 + 2\ FeCl_2 + 2\ HCl \\ \hline S^o + 6\ FeCl_3 + 4\ H_2 \longrightarrow H_2SO_4 + 6\ FeCl_2 + 6\ HCl \end{array}$$

T. thiooxidans has been reported to oxidize elemental sulfur to sulfate under anerobic conditions and *T. denitrificans* oxidizes elemental sulfur to sulfate anerobicly in the presence of nitrate. Under natural conditions, these oxidations may be mediated by the sulfate reducing bacteria which have been reported to reduce elemental sulfur to sulfide.

Oxidation of elemental sulfur by ferric iron has also been shown to occur in the presence of *T. thiooxidans* and *Sulfolobus acidocaldarius*. Oxidation of elemental sulfur and reduced sulfur compounds has also been shown to occur in the presence of *T. denitrificans* in the presence of nitrate. Thus, it can be seen that under certain conditions sulfur can be oxidized to sulfuric acid in the absence of oxygen.

Sulfate-reducing bacteria reduce elemental sulfur and oxidized compounds of sulfur to sulfide by way of sulfite. Sulfite, when produced in excess, can combine with elemental sulfur to form thiosulfate. Should this thiosulfate migrate to regions in which free oxygen is available, the sulfur oxidizing bacteria can oxidize this compound with the production of sulfuric acid.

FORMATION OF JAROSITES

Secondary iron minerals are widespread in nature: ochre deposits, "yellow boy" deposits along the banks of acidic streams, and crusts and coatings on ferruginous ores and in cracks of adjoining rocks. The principle minerals responsible for these conditions are known as jarosites. Although formed by abiological reactions, the occurrence of these minerals is intimately associated with biological iron oxidation. In the presence of suitable cations, ferric sulfate is hydrated to produce jarosites according to the following equation:

$$3\ Fe_2(SO_4)_3 + 2\ AOH + 10\ H_2O \dashrightarrow 2\ A[Fe(SO_4)_2.2\ Fe(OH)_3] + 5\ H_2SO_4$$

Of the nine known jarosite-type compounds, six are found in nature: these are jarosite ($A = K$); natrojarosite ($A = Na$); argentojarosite ($A = Ag$); ammoniojarosite ($A = NH_4$); plumbojarosite ($A = Pb$); and hydronium jarosite or carphosiderite ($A = H_3O$). A lead and copper containing member of this group, beaverite is also known. Of all the jarosites, the potassium-containing mineral is the most stable, and in the presence of potassium ions the other jarosites will be converted to this mineral. The interconversions of the jarosites and their solubilities are controlled by the prevailing chemical and physical conditions especially with regard to the concentrations of iron, monovalent ions and sulfate, the temperature and pH. In this regard, the following generalizations can be made (Silver *et al*, 1986). After potassium jarosite, the most stable species are, in order of stability, the sodium, ammonium and hydronium jarosites; thus the greater the proportion of monovalent ions being potassium, the greater the precipitation of the jarosites. Increasing the concentrations of the monovalent ions up to the stoichiometric limits of the reaction will also favour precipitation. Increasing temperature, and increasing alkalinity favours jarosite formation, but pH values much greater than two will result in iron precipitates other than the jarosites, such as goethite. The optimum pH for jarosite formation is between 1.8 and 2. Below pH 1.8, jarosite forms very slowly if at all, or tends to dissolve. The genesis of jarosites causes the removal of ferric iron and monovalent ions from solution in a disproportionally greater amount than the sulfate; thus the concentrations of free sulfate ions tends to increase with a corresponding decrease in pH. Monovalent ions necessary for jarosite formation may come from dissolved monovalent ions in solution or from monovalent ions liberated from feldspar minerals. Environmental consequences of jarosite formation are: the removal of iron from solution; the coating of mineral surfaces by jarosites thus inhibiting mineral oxidation; and the relative increase in sulfate ions and thus the lowering of pH.

FACTORS AFFECTING MINERAL OXIDATION

The reactions involved in the alterations of minerals by bacteria are predominately chemical in nature. These, like all other chemical reactions, are controlled by a number of chemical and physical factors which influence the rates of these reactions and thus determine the extent of acid generation and heavy metal liberation.

Temperature

Although various bacteria have been shown to grow between temperatures of -18°C and 105°C, the transformation of minerals by microorganisms are confined to a more limited range. Most physiological studies with the iron oxidizing thiobacilli have indicated optimal temperature between 28°C and 35°C, with the maximum being between 40°C and 45°C. The lower temperature limits of the iron oxidizing thiobacilli have not as yet been established but bacterial iron oxidization in mines, soils and natural waters have been noted below 10°C; the lower limit of this activity is generally considered to be near the freezing point of water. Thermophillic iron oxidizing bacteria, both rod shaped and lobate spherical cells have been isolated from sulfide mineral bearing environments that might represent a natural succession to the iron oxidizing thiobacilli as localized temperatures increase due to the exothermic oxidation of pyrite. These possess optimum temperatures between 50 and 70°C with maxima of 50-90°C.

The rates of biological reactions increase with increasing temperatures until a maximum velocity is attained; further increases in temperature result in decreases in the reaction rate. To compare the rates of biological reactions ratio of the velocity of these reactions, a predetermined temperature interval, usually 10°C is used and is designated by the symbol Q_{10}. Q_{10} values for several of the chemical activities of iron oxidizing bacteria have been calculated and are without exception approximately two. That means that for each ten degree increase in temperature, the rate of reaction doubles.

Acidity.

Like most living organisms, the iron oxidizing thiobacilli exhibit pH optima for growth and for their various metabolic activities. These, however, are not necessarily coincidential. Cultures of the iron oxidizing bacteria can grow and oxidize ferrous iron well between pH 1.5 and 5 but may remain active at pH values as low as 1 or as high as 6.5. The pH may change during growth and oxidation of iron compounds and minerals due to the nature of the reactions. For instance, in the initial stage of the oxidation of ferrous sulfate, sulfate is consumed in the reaction

$$4\ FeSO_4 + O_2 + 2\ H_2SO_4 \rightarrow 2\ Fe_2(SO_4)_3 + H_2O$$

but is generated in the hydration to ferric hydroxide

$$Fe_2(SO_4)_3 + 6\ H_2O \rightarrow 2\ FeOH_3 + 3\ H_2SO_4$$

or hydration to form secondary iron sulfate minerals such as jarosite

$$3\ Fe_2(SO_4)_3 + 12\ H_2O \rightarrow 2\ H[Fe(SO_4)_2.2Fe\ (OH)_3] + 5\ H_2SO_4$$

The oxidation of elemental sulfur and such metal sulfide minerals as pyrite, marcasite and arsenopyrite results in the formation of sulfate with a corresponding decrease in pH, whereas the oxidation of pyrrhotite, chalcosite, digenite and chalcopyrite uses sulphate and thus cause decrease in the acidity. The formation of insoluble sulfates such as anglesite ($PbSO_4$) or antimony sulfate ($Sb_2(SO_4)_3$) also removes sulfate from solution and thus tends to increase the pH. The formation of the secondary copper mineral, antlerite

$$3\ CuSO_4 + 4\ H_2O \longrightarrow Cu_3(SO_4)(OH)_4 + 2\ H_2SO_4$$

also removes sulfate ions from solution and thus tends to increase the pH. The formation of jarosite and antlerite are stimulated in response to decreasing acidity.

During the formation of ferric iron at pH values greater than 1.7, the metabolic activity of the iron oxidizing thiobacilli create conditions which encourage the precipitation of secondary minerals of iron such as jarosite. This phenomenon influences metal sulfide mineral oxidation in two ways: not only is a leaching agent, ferric iron removed from solution but the precipitates formed insulate the mineral crystal surfaces from the attack of both the ferric iron and the bacteria.

Oxidation-reduction potential.

The oxidation reduction potential (Eh), also called the electromotive force (emf), redox potential or O/R potential, is a measurement of the ability of a substance to accept or donate electrons, and is used to measure the oxidizing ability of a given environment. Each compound or element that can be oxidized or reduced has its own characteristic redox potential which, when measured against a standard substance, can be classified in an electromotive series and assigned an numerical value. The strongest reducing agents would have larger negative values whereas the strongest oxidizing agents would have larger positive values. Simplified Eh/pH diagrams as presented in Figure 3 can illustrate the stability relationships of different species, describing conditions at which the species will tend to be stable or will undergo modifications. Using this information, it would then be possible to predict the alteration of minerals under certain conditions. Using the Eh/pH diagram for iron, we can see that under conditions more acid then that defined by the heavy line between $Fe(OH)_3$/siderite and $Fe(OH)^{2+}/Fe^{2+}$, iron would tend to dissolve. Except under very acid and or oxidizing conditions, most of the iron in solution would be more stable in the ferrous valency. Above pH 1.7, hematite and other iron precipitates would tend to form; in the presence of sulfate ions as opposed to the bisulfate ions at more acid conditions, this would stimulate the formation of iron sulfate secondary minerals such as jarosite. Iron sulfide minerals would also decrease in stability and therefore dissolve more readily under more oxidizing conditions as which a

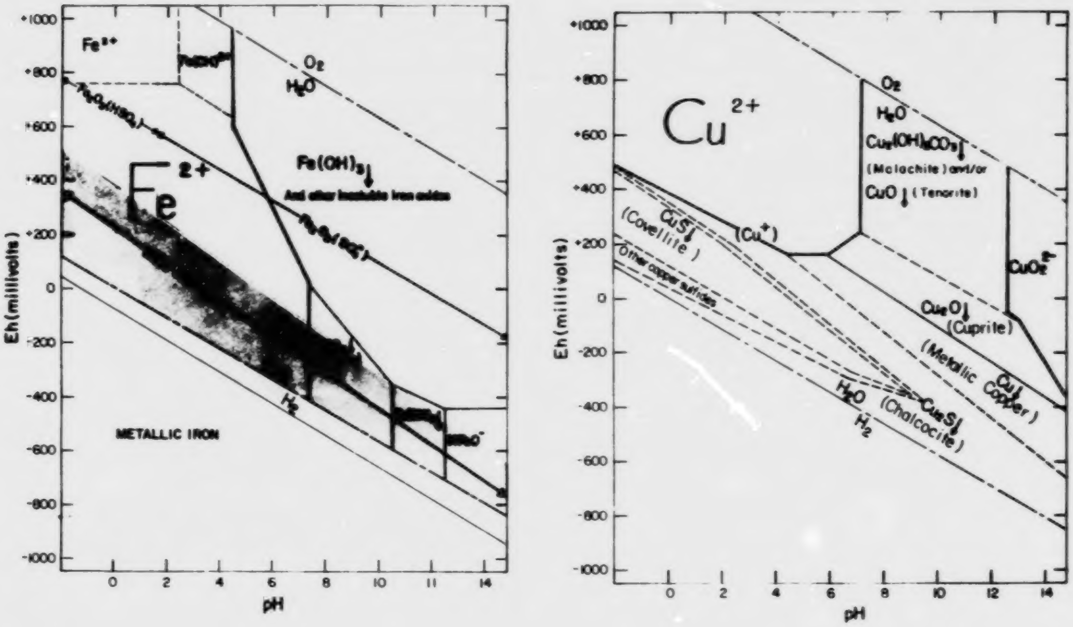

Figure 2. Eh/pH diagrams for iron and copper.

greater proportion of the iron ions are in the trivalent state. Like iron, the stability of soluble ionic copper is clearly defined by the conditions more acid or more oxidizing then indicated by the area between the heavy lines in the diagram. Divalent cupric ion is much more stable over a wide range of conditions than the univalent cuprous ion. When conditions are more acid and more oxidizing than those indicated by the heavy line, copper minerals tend to dissolve releasing cupric ions into solution. The acidity may be caused by bacterially-formed sulfate ions; the oxidizing conditions may be caused by high concentrations of ferric iron. Upon the oxidation of the copper minerals, ferric iron is reduced to the ferrous state and the Eh will decrease unless the iron is reoxidized.

During the oxidation of ferrous iron by iron-oxidizing bacteria, the oxidation/reduction potential might be expected to increase toward that of the ferric/ferrous system, which is +747 mV. In reality this value is rarely approached except at very low iron concentrations and of at high acidity as oxidizing iron is precipitated from the medium as jarosite. EH values between +200 and +550 mV have been measured during this oxidation, or during the oxidation of metal sulfides. Similar pH values have been found for pore water in tailings under oxidizing conditions or in effluents from uranium tailings; upon covering the tailings with vegetation and arresting iron oxidation, Eh values have been observed to decrease to between +100 to -200 mV.

Oxygen Concentration

Oxygen concentration is one of the limiting factors in bacterial oxidation of minerals. Myerson (1981) showed that the concentration of dissolved oxygen below which oxygen acts as the limiting factor is 0.5 mg/L. The affects of gas phase oxygen concentration on bacterial pyrite oxidation has been investigated by a number of investigators, who found that the concentration at which oxygen became limiting ranged between 2 and 20% (Erikson, 1985).

Although research in uranium tailings (Harries and Ritchie, 1984) and coal spoils (Erikson, 1985) has shown that the oxygen concentration decreases rapidly from 20% at the surface to between 1 and 5% within a few meters of the surface, pyrite oxidation occurs at much greater depths. Examination of bacterial populations in American copper ore leaching operations (Beck, 1967; Bhapppu *et al*., 1969; Murr, 1980), Australian uranium tailings (Babij *et al*., 1981, Goodman *et al*., 1981) and coal spoils (Schubert and Miller, 1982) showed that these bacteria exist at high concentrations near the tailings surface. More recent studies on the bacterial populations of the Nordic uranium tailings showed that the iron oxidizing bacteria are found at or just below the surface and at the water table/capillary fringe of the tailings (Silver, 1987) possibly explaining the reason why surface vegetation fails to arrest sulfuric acid generation (Durbrovsky *et al*, 1985). Various studies have shown that bacterial pyrite oxidation is significantly decreased by flooding of various tailings: pyritic uranium tailings (Halbert *et al*., 1982, Ritcey and Silver, 1982); underground coal mine workings (Ladwig *et al*., 1984); and base metal tailings (Hamilton and Frazer, 1978). Erikson (1985) showed that covering pyritic spoils with inert cover materials such as clays may not be useful in decreasing bacterial pyrite oxidation, but instead an actively oxygen consuming layer such as vegetated soil would be necessary. This concept was confirmed when Silver and Ritcey (1985) showed that bacterial pyrite oxidation in uranium tailings could be arrested by the application of oxygen consuming compost placed upon the surface of pyritic uranium tailings. Laboratory research on a zinc sulfide ore (Goodman et al., 1983) showed that decreasing the oxygen concentration by nitrogen, methane, or carbon dioxide are also effective in inhibiting bacterial sulfide mineral oxidation. These gasses are synthesized by bacterial action in wetlands (Kaplan et al, 1979; Valiela, 1984).

Hydrology

Water is a principle component of environment in which bacterial sulfide mineral oxidation occurs, functioning as a reactant, a reaction medium, a transport medium for bacteria, nutrients and metabolic products, and an oxygen barrier. Hydrology is thus one of the most important factors to consider in in situ bacterial sulfide mineral oxidation (Ladwig, 1985). Description of the hydrology of an area defines the recharge and subsurface water flow rates and volumes, lateral variations in water quality, and contaminant loadings. An understanding of the hydrology is important to derive the interaction between acid production and hydrologic transport and in defining the interaction of the constiuents of subsurface water and the hydrochemical evolution of this water.

Redox values of ground water are determined by the relative rates of introduction of oxygen and the oxidation of organic compounds and sulfide minerals (Drever, 1982). The most important variables in natural systems appear to be: the oxygen content of the recharge water; the distribution and reactivity of organic and inorganic substrates for aerobic bacteria; the distribution and concentrations of potential redox buffers such as $Fe(OH)_3$; and the circulation of the ground water. Ground water located near the surface or in the water table/capillary fringe area tends to have a high redox potential and dissolved oxygen. In the presence of oxidizable substrates, oxygen concentrations and redox potential decrease with depth. Oxygen concentration and redox potential may also vary with horizontal movement of ground water, for example when oxygenated water flows through an aquifer containing oxidizable substrates. This redox front will move in the direction of ground water flow but at a much slower rate than the ground water.

EFFECT OF PREVENTION OF OXYGEN INDUCTION INTO TAILINGS

Ritcey and Silver (1982) and Silver and Ritcey (1985) have shown that exclusion of oxygen from pyrite containing uranium tailings completely prevented sulfuric acid generation. The influx of oxygen was prevented by either a layer of water, or by a layer of decomposing cellulosic compost. Sulfide mineral oxidation would be expected to be inhibited in base metal tailings when the influx of oxygen is prevented by a layer of vegetated soil.

CONCLUSIONS

Bacterial oxidation of elemental sulfur and sulfide minerals is the primary cause of acid generation in base metal tailings. Factors that influence bacterial sulfide mineral oxidation are pH, bacterial activity, temperature, oxidation reduction potential and oxygen availability. Decreasing oxygen availability by the application of oxygen consuming materials on the surface of the tailings should decrease bacterial sulfide mineral oxidation except when the tailings contain appreciable concentrations of elemental sulfur. Hydrological aspects including geohydrochemistry are important in determining the interaction between bacterial metal sulfide oxidation and the environmental consequences.

Figure 3 shows the comparison in events between a hypothetical uranium or base metal tailings deposit in which oxygen induction and seepage exit are unimpeded, and one in which the tailings are isolated from the atmosphere and ground water by organic material. In the first case, oxygen enters the tailings and can thus serve as oxidant for sulfide mineral oxidation which generates acidity in the form of sulfuric acid and solublizes heavy metals. In the case of uranium tailings, radon gas (generated by the nuclear disintegration of radium), heavy metals and radionuclides can migrate to the surface of the tailings. Sulfuric acid, soluble heavy metals and radionuclides can also enter the ground water from the tailings beneath the surface. When vegetation is placed on the surface of the tailings, oxygen entry into the tailings is prevented and thus metal sulfide mineral oxidation is arrested. Radon gas and soluble radionuclides and heavy metals are trapped in the soil or organic layers. Depositing the tailings on a layer of organic matter such as peat moss, or by placing vegetated wetland soil downstream from the tailings (Pesavento, 1987; Wieder and Lang, 1984) removes sulfate and heavy metals from the seepage. Thus an ideal situation can be envisaged in which organic materials such as compost, vegetated soil or peat moss can effectively isolate the environment from deleterious chemical and biological events in uranium and base metal tailings.

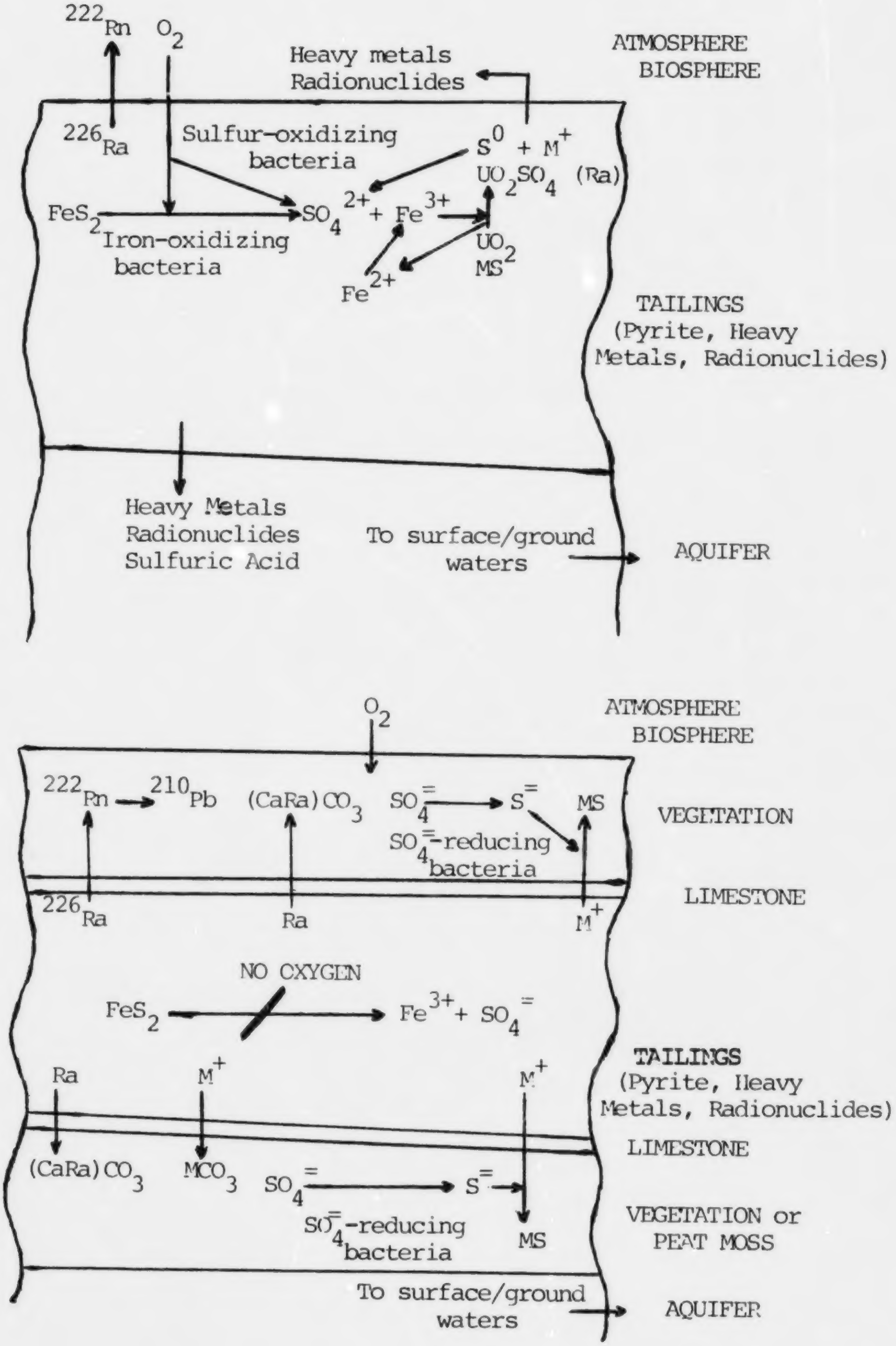
222Rn
O2
Heavy metals
Radionuclides
ATMOSPHERE
BIOSPHERE
226Ra
Sulfur-oxidizing bacteria
S0 + M+
UO2SO4 (Ra)
FeS2
SO4 2+ + Fe3+
Iron-oxidizing bacteria
UO2
MS2
Fe2+
TAILINGS
(Pyrite, Heavy Metals, Radionuclides)
Heavy Metals
Radionuclides
Sulfuric Acid
To surface/ground waters
AQUIFER
O2
ATMOSPHERE
BIOSPHERE
222Rn
210Pb
(CaRa)CO3
SO4=
S=
MS
SO4=-reducing bacteria
VEGETATION
226Ra
Ra
M+
LIMESTONE
NO OXYGEN
FeS2
Fe3+ + SO4=
TAILINGS
(Pyrite, Heavy Metals, Radionuclides)
Ra
M+
M+
LIMESTONE
(CaRa)CO3
MCO3
SO4=
S=
SO4=-reducing bacteria
MS
VEGETATION or PEAT MOSS
To surface/ground waters
AQUIFER

Figure 3. Comparison of hypothetical metal sulfide mineral containing tailings deposit from the environment with organic materials.

ACKNOWLEDGEMENTS

The concepts described were developed while the author was employed at CANMET, Energy Mines and Resources Canada, Ottawa, to which he expresses his appreciation. Special thanks are due Mr. Gordon Ritcey for his unstinting support, leadership and collaboration.

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LUNCHEON ADDRESS

The Honourable Laird Stirling
Minister of the Environment
Nova Scotia

Mr. Chairman, head table guests, ladies and gentlemen.

I would like to thank you for inviting me to address this timely and important seminar.

The organizing committee and OceanChem Group Limited are to be commended for preparing a comprehensive programme touching on many, if not all, the various aspects of Acid Mine Drainage. I am pleased to note the contributions of speakers from across Canada and the United States and on behalf of the Government of Nova Scotia, would like to welcome them to our province.

It has been suggested that I spend some time speaking to the quasi-technical aspects of Acid Mine Drainage. I have glanced through your programme and note that over the past day and one half you have examined Case Studies, pondered the Identification of Sources and Impacts, speculated on Predictive Methodologies and grappled with Prevention. I would be somewhat out of my area of expertise in attempting to summarize those issues or this afternoon's issues related to Control and Treatment of Acid Mine Drainage. I think that it is sufficient to recall that acid drainage resulting from mineralized slates of the Halifax Formation has been identified as the cause of major fish kills and disruption of public and private water supplies in certain areas of Nova Scotia.

Since I have been Environment Minister in Nova Scotia for only a very short period of time, my familiarity with Acid Mine Drainage may be termed marginal. However, I am somewhat more familiar with the complexities of acid precipitation. One of my first tasks as Minister of the Environment was to make a presentation to the House of Commons Special Committee on Acid Rain. During the process of preparing for that presentation I was introduced to the sources, environmental effects and treatment technologies applicable to acid rain. I learned that many of Nova Scotia's lakes and streams are unable to neutralize acidity and that acid mine drainage is another source of acidity that can be harmful to our water resources.

In my opening remarks I noted the timeliness of this seminar. I made this observation based on Nova Scotia's on-going commitment to develop fully and utilize our natural resources. The speech from the Throne delivered by His Honour the Lieutenant Governor a few weeks ago contained this statement:

> "... my Government will work to enhance the natural resource sector of our economy through the identification of goals for resource development and the implementation of stragegies to enable us to attain those goals."

One of the strategies entails

> "Protecting and enhancing Nova Scotia's environment in all its aspects".

Our goals include a management plan for inland fisheries and increased efforts for the development of our Tourism potential. At the same time we are actively encouraging further mineral exploration and development projects and increased utilization of Nova Scotia Coal. Without proper environmental considertation these goals may not be realized. That is, the negative impacts which can result from mining and similar activities, including disruption of slates for construction purposes, could interfere with development of an inland fishery and our ability to attract tourists.

Our goal in Nova Scotia is to create long-term economic development. It will not be possible however to sustain development without a strategy that incorporates environmental protection and conservation measures. Frankly, little benefit can be gained from pursuing developments which create employment today but, because environmental safeguards are not provided, foreclose on future development opportunities. As a member of the National Task Force on Environment and the Economy I am aware of the need to promote and incorporate environmental values in economic development planning. I am confident that this view is gaining ground in both the private and public sectors. The recent Speech from the Throne and the interest in the National Task Force on Environment and the Economy shown by government, industry and interest groups are examples of this growing interest.

Maintaining our environment so that we can continue to develop economically and socially is, as I have already implied, a shared responsibility of government, industry and the public at large. An important aspect of this partnership is the requirement to research, assess and exchange information on problems such as acid mine drainage which can impede the realization of our environmental and economic development goals. Your seminar will play an important role in this information transfer process. My hope is that it will set the stage for the development, refinement and implementaion of skills and techniques for managing acid mine drainage so that the commitments made in our Government's Speech From the Throne can be realized for the benefit of Nova Scotians.

Thank you Mr. Chairman for the opportunity to speak to you on this very important topic and to extend my wishes for a successful conclusion to this seminar.

Acid Mine Generation at
NB Coal Ltd's Fire Road Operation
Minto, N.B.

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Abstract

This report considers the sources, treatment and plans for the minimization of acid minewater generation at one of NB Coal Ltd.'s cut and fill coal mining operations.

The unexpected requirement to neutralize minewater required the rapid set-up of a system capable of treating 400 to 900 m^3/h of minewater. This was followed by an investigation which confirmed waste rock as the source of acid generation and the consequent development of plans to minimize the generation of acidic drainages.

Acknowledgement

I would like to thank Mr. A. Cormier, President, NB Coal Ltd., for authorization to prepare this report for presentation at the Acid Mine Drainage Seminar in Halifax, Nova Scotia (March 1987).

1.0 INTRODUCTION

This report considers the origin, treatment and plans for the minimization of acidic minewater associated with one of NB Coal Ltd.'s coal mining operations near the town of Minto in central New Brunswick.

Approximately 50,000 tonnes per annum of thermal coal are extracted from NB Coal's Fire Road operation. Cut and fill mining methods are used to recover the thin seams of coal which are overlain by as much as 20 m of bedrock. The general layout and approximate extent of the operation are shown in Figure 1-1.

The mine has been in operation since late 1983. Mining proceeds in an easterly direction starting at the mine road shown in Figure 1-1. Water collected in the cut drains to the western limit of the cut from where it is pumped into a series of sedimentation "ditches". This minewater is a combination of groundwater and surface runoff. The groundwater tends to enter the cut by "bubbling" up through the exposed coal seam. Surface runoff enters the cut primarily from the drainage area of approximately 410 ha to the south of the mining operations (Figure 1-1). A diversion ditch was constructed to convey most of this drainage away from

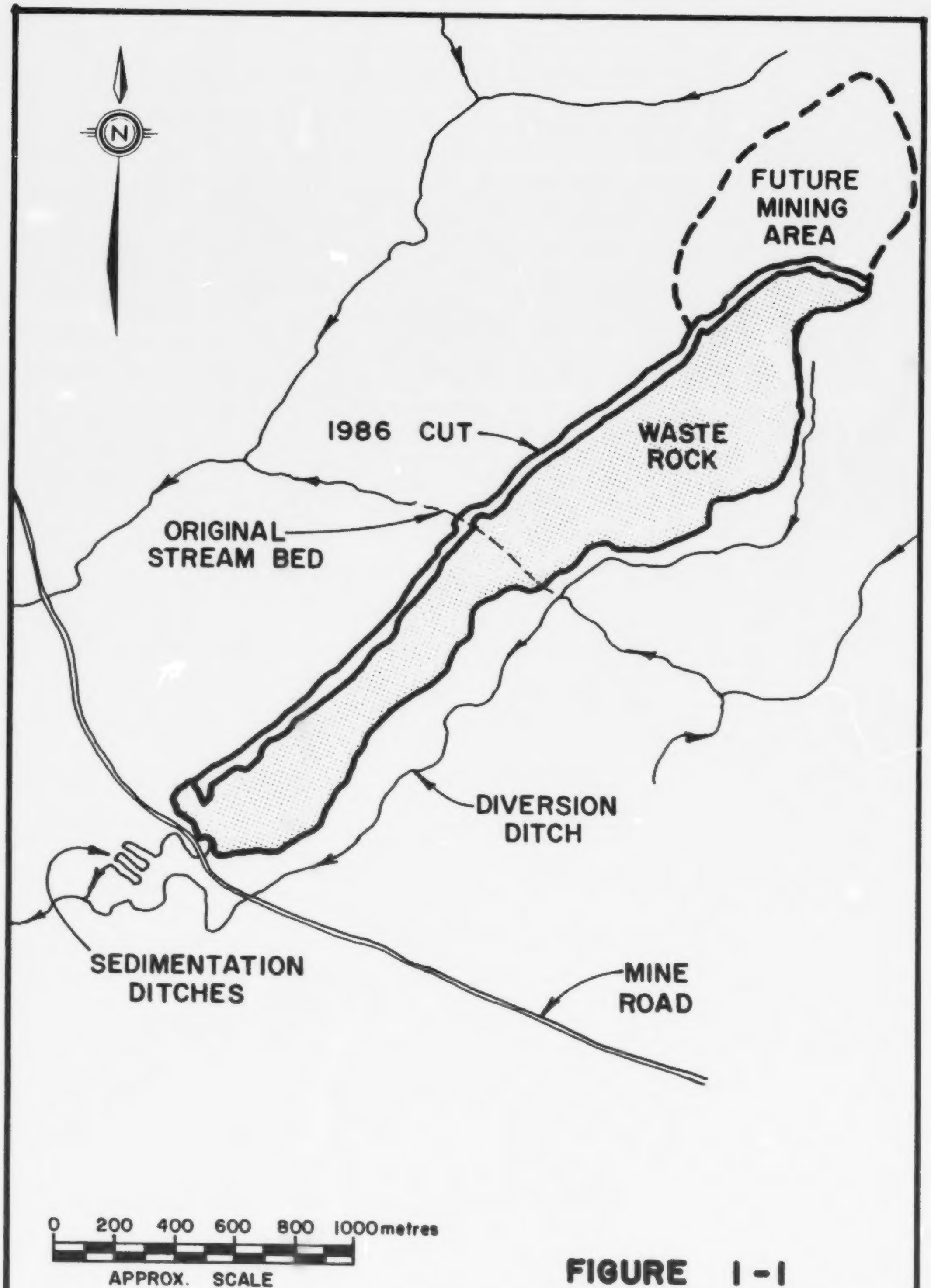

FUTURE MINING AREA
1986 CUT
WASTE ROCK
ORIGINAL STREAM BED
DIVERSION DITCH
SEDIMENTATION DITCHES
MINE ROAD
N
0 200 400 600 800 1000 metres
APPROX. SCALE
1: 20,000

FIGURE 1-1
NB COAL LTD
FIRE ROAD OPERATION

the cut. However, surface water forms pools all along the southern limit of the waste rock disposal area. These form a probable source of seepage through the waste rock into the cut.

During the initial mining phase, the pH of minewater was in the range of 4.8 to 7.4. However, after approximately one year of operations the pH decreased to a range of approximately 3.0 to 4.0. The discharge of this minewater, at flowrates of approximately 400 to 900 m^3/h, was regarded as unacceptable and hence urgent corrective action was necessary.

2.0 INITIAL TREATMENT PHASE

The pumping of water from the cut was discontinued in May 1986 due to environmental concerns. Unfortunately the level of water in the cut was such as to cause potential flooding of the mining operation. Hence, it was desirable to recommence pumping as soon as possible to avoid shutdown of the coal mining operation.

The cut contained approximately 64,000 m^3 of water at an acidity, measured to pH 8.0, of approximately 400 mg/L as $CaCO_3$. Visual observations indicated that a large quantity of water was entering the cut from the upper or eastern end (Figure 1-1). This inflow was characterized by an acceptable pH in the range of 6 to 7. Thus, it appeared feasible to "quickly" increase the pH of the minewater by the addition of a soluble alkali, such as caustic soda. On this basis approximately 34 tonnes of caustic soda (50 wt%) were added to the cut and the total acidity and pH monitored for several days following the addition.

The acidity of the minewater decreased substantially following the addition of caustic soda to the pool of minewater (Table 2-1). However, alkalinity material balances indicated that the continued influx of acidity into the cut was

TABLE 2-1

Minewater Quality

	24 May 86	10 Jun 86	20 Jun 86	3 Jul 86	7 Jan 87
Turbidity (NTU)	62			27	-
pH	3.05	4.05	4.19	3.80	2.9-3.2
Acidity to pH (mg/L as $CaCO_3$)					
5.0	255				
6.0	310	53.7	39.3	39.0	
6.5					623-1030
7.5		74.1	53.7	51.2	713-1120
8.0	408				
8.5		84.7	61.2	57.1	788-1220
9.5	513				
Extractable Metals (mg/L)					
Al	34.3	10.3	7.3	5.9	84.5-134
Fe	41.2	5.48	4.54	0.53	91.2-134
Mn	18.6				
Ca	96.5	86.5	6.05		
Mg	29.2	13.2	10.0		
Cu		0.05	0.03		
Zn		0.52	0.41		
Sulphate (mg/L)	790	468	290		

considerable. This, combined with the poor mixing dynamics of the long, narrow pool of minewater, made the further addition of caustic soda impractical as a means of achieving an acceptable pH.

During the one week period of caustic soda addition, arrangements were completed to enable continuous treatment of the minewater while pumping from the cut at flowrates of approximately 400 to 800 m^3/h. Approximately one tonne per day of powdered hydrated lime was added on a continuous basis to the suction side of one of the two minewater pumps. The length of 150 mm diameter discharge pipe was approximately 100 m. The mixing provided by the pump and discharge pipe was very adequate to dissolve sufficient lime for neutralization of the total flow from both minewater pumps. Neutralization and the oxidation of ferrous iron were completed in a 100 m long ditch immediately downstream of the discharge pipes. The exit of the pipes was arranged to provide splashing of the discharge to maximize re-aeration of the water for complete oxidation of iron. The rate of lime addition was "crudely" controlled by hourly monitoring of pH at the discharge end of the ditch. Summer students were used for both the addition of lime and monitoring operations.

The aforementioned ditch discharged into a series of sedimentation ditches. These provided a nominal clarification surface area of approximately 3000 m^2. However, the effective area was probably significantly less than this due to the velocity of flow through these ditches.

The continuous treatment of minewater was carried out during the months of June and July 1986. By the end of July the cut had been nearly emptied of water. The inflow of water into the cut was relatively slow at this time due to the summer low-flow runoff conditions. These factors, combined with NB Coal's decision to discontinue coal mining operations at the Fire Road site, led to the decision to at least temporarily end the treatment operation.

The continuous treatment method proved to be adequate as indicated by the results of effluent monitoring (Table 2-2) and a 96-hour acute toxicity bioassay test (100% survival in 100% effluent).

TABLE 2-2

Fire Road - Sedimentation Ditch Overflow

June - July 1986

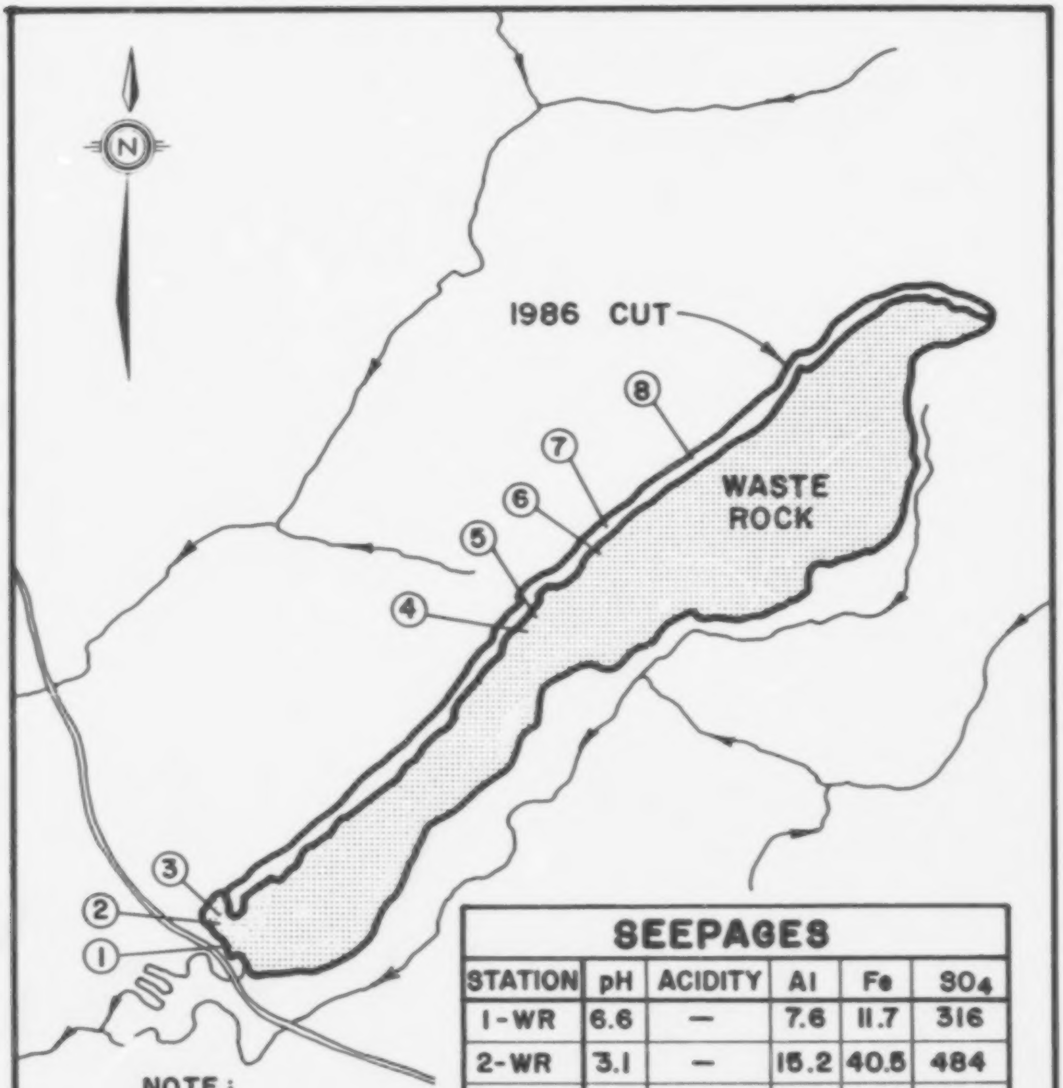
	Turbidity NTU	Suspended Solids mg/L	pH
Number of Analyses	42	42	
Minimum	3.8	4	4.5
Maximum	62	52	11.0
Median	12	16	7.0
10 Percentile	5.3	6	-
90 Percentile	28	33	-

3.0 SOURCE OF ACID GENERATION

Both the coal and overburden sandstones contain iron sulphides. The thin seams of coal are removed almost completely. Residual coal is largely flooded by water accumulated in the cut. Thus, coal was not considered to be a major source of acidic minewater. A much more probable source was the extensive areas (approximately 30 ha) of waste rock located between the cut and the surface water stream along the southern limit of the waste rock dump.

The waste rock dump, which rises 10 m and more above the landscape, provides an ideal environment for the acid generation process. Infiltration of air (oxygen) and moisture (or humidity) into the dump would occur very readily due to the coarseness of the waste rock (i.e. run of mine material). Periodic flushing of the products of acid generation into the cut would be enhanced by the seepage of surface water through the waste rock into the cut from the surface water stream and pools along the southern toe of the dump (Figure 1-1).

A survey of visible seepages, following the lowering of the level of water in the cut, confirmed the waste rock dump as the probable primary source of acid generation. The results of the survey are given in Figure 3-1. All of the seepages from the waste rock were acidic to varying degrees. The flow of water along the high wall side of the upper end of the cut did not indicate the presence of acid generation. This water consisted primarily of groundwater emitted into the cut through the coal seam.



SEEPAGES					
STATION	pH	ACIDITY	Al	Fe	SO_4
1-WR	6.6	—	7.6	11.7	316
2-WR	3.1	—	15.2	40.5	484
3-WR	3.3	—	19.6	6.49	456
4-WR	6.0	—	2.6	6.70	87.0
5-WR	3.2	298	32.8	8.91	710
6-WR	2.8	607	106	109	2000
7-POOL	2.7	2890	260	306	4520
8-GW	7.7	—	0.10	0.40	20.6

NOTE:

ACIDITY – TO pH 8.5 AS $CaCO_3$
Al, Fe, SO_4 – mg/l
WR – SEEPAGE FROM WASTE ROCK
POOL – POOL OF WATER AT TOE OF WASTE ROCK
GW – GROUNDWATER FLOW ON HIGHWALL SIDE OF CUT

0 200 400 600 800 1000 metres
APPROX. SCALE
1:20,000

FIGURE 3-1
SOURCES OF
ACID GENERATION
NB COAL LTD
FIRE ROAD OPERATION

4.0 FUTURE MANAGEMENT OF THE AREA

The cut was allowed to flood as of August 1, 1986. The level and pH of the water in the cut were monitored on a routine basis. Experience at other abandoned cuts in the region suggested that the level of water in the cut might stabilize well below the overflow point. This was an acceptable approach to the provincial regulatory agency providing no adverse environmental effects were observed downgradient of the cut. However, by late December it was apparent that pumping and treatment of water would be required to prevent uncontrolled overflow from the cut.

Pumping and treatment of minewater were initiated early in January 1987. As shown in Table 2-1, the acidity of the minewater, recorded in January, was substantially higher than during the earlier treatment operations. This required the addition of approximately 8 tonnes per day of hydrated lime for the neutralization of approximately 400 m^3/h of minewater. Hydrated lime was added as a slurry to the suction side of the one minewater pump in operation. The slurry was prepared by the addition of bag lime to an $8m^3$ tank. The tank was provided with a recirculation pump for mixing. A hose from the discharge side of the recirculation pump was used to deliver slurry to the suction side of the minewater pump. Periodic measurements of pH were used to adjust the rate of lime addition. This system was successful in maintaining the required effluent pH (6.5 to 8.5) as indicated in Table 4-1. However, some post-precipitation of ferric hydroxide was observed in samples of the treated effluent for which the pH was less than 7. This can be attributed to the relatively slow rates of oxidation of ferrous iron under winter conditions. In future, the pH of the minewater will be raised to 8.0 or greater to ensure that this factor does not limit the rate of oxidation.

The amount of freeboard in the cut had increased to an acceptable level by the end of January 1987.

The economic costs associated with the acid generation problem are a factor in NB Coal's decision to cease mining operations at the Fire Road site. As indicated in Figure 1-1, further mining operations will be limited to the recovery of coal which is close to the surface at the eastern end of the cut. This operation is expected to be completed by June 1987.

Various measures will be undertaken, likely in stages, to minimize the effects of acid generation. The diversion

TABLE 4-1

Fire Road - Sedimentation Ditch Overflow
January 1987

	pH
Number of Analyses	32
Minimum	4.3
Maximum	9.3
Median	7.5
10 Percentile	5.5
90 Percentile	9.0

ditch along the southern limit of the waste rock dump has been deepened in order to better drain the surface runoff which tends to pool along the dump. The cut will be filled with waste rock and the whole area then contoured to ensure the most rapid runoff of surface drainage. The results of these efforts will be monitored prior to decisions concerning further action. It may be necessary, in the longer term, to cap the area with impermeable material to minimize both the access of air and the rate of infiltration of surface runoff into the waste rock. In the meantime, the temporary treatment facilities will be used, on an as required basis, to produce an acceptable treated effluent.

NB Coal has also initiated efforts to better predict the potential for acid generation during exploration and development programs. This will enable the earliest possible consideration of the potential for and implications of acid generation.

ENVIRONMENTAL & RECLAMATION MEASURES AT EQUITY SILVER MINES LTD.

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Houston, British Columbia

January, 1987

ABSTRACT

Acid Mine Drainage (A.M.D.) formed from oxidation of disposed pyritic wastes presents a potential pollution hazard for Equity Silver Mines Limited and the mining industry in general. Failure to collect and treat these acidic effluents can eventually affect the quality of receiving streams and endanger aquatic life forms. Treatment and discharge of A.M.D. to the receiving environment must comply with stringent anti-pollution laws.

Wastes mined at Equity Silver Mines Ltd. are predominantly acid generating and must be handled according to a materials management plan to minimize impact on the environment. Effluent collection and treatment facilities have been constructed to address the short term environmental concerns with reclamation and research programs being developed to address and monitor abatement measures for control over the long term.

The purpose of this paper will be to offer a brief overview of these measures. Key topics for discussion will be: kinetics of Acid Mine Drainage, acid generating potential, water treatment and sludge handling, special wastes, waste dump construction, waste rock amendments, bacteriocide research, simulations, and reclamation measures. Programs designed to evaluate impact on the receiving environment, although they form an extensive part of the program, will not be discussed in this paper.

INTRODUCTION

Equity Silver Mines Limited is located in the Central Interior of British Columbia, within the Omineca Mining Division, 35 kilometers southeast of Houston and approximately 575 kilometers by air north northwest of Vancouver (Figure 1).

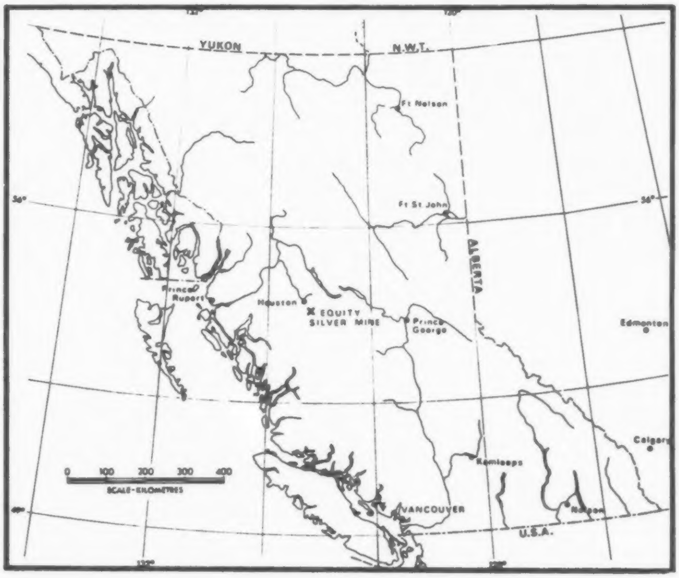
YUKON
N.W.T.
Ft Nelson
Ft St John
ALBERTA
Prince Rupert
Houston
EQUITY SILVER MINE
Prince George
Edmonton
Calgary
Kamloops
VANCOUVER
Nelson
U.S.A.
0 100 200 300 400
SCALE-KILOMETRES

FIGURE 1: LOCATION OF EQUITY SILVER MINES LTD.

The deposit consists of three economic mineralized zones: The Southern Tail, Main and Waterline Zones. Mining of the Southern Tail commenced in April, 1980 and was completed in the first quarter of 1984. Main Zone production was initiated during the last quarter of 1983 and the Waterline Zone remains undeveloped to-date. Remaining reserves are in the order of five to six years contingent on favourable metal prices.

The three deposits (Figure 2) occur within a window of interbedded volcanic and minor sedimentary rocks. The older volcanic sequence is bounded on the west by a small granite-like stock and on the east by a gabbro-monzonite complex. All are overlain by a series of younger lava flows.

Economic mineralization (Cu-Ag-Au) occurs as tetrahedrite and chalcopyrite. Pyrite is the predominant sulphide mineral throughout the waste and mineralized zones and occurs in disseminated, massive and crystalline forms. Oxidation of these sulphides results in the formation of Acid Mine Drainage (A.M.D.). Although reaction mechanisms are complex, the key parameters have been identified as moisture, oxygen, a source of sulphide minerals and the presence of certain strains of iron and sulphur oxidizing bacteria. (Kleinman & Erickson, 1983). Elimination of any one of these parameters will suppress the acid producing reactions.

Equity Silver Mines Ltd. has and is continuing to encapsulate wastes with glacial till in an effort to eliminate control factors governing the rate of oxidation and subsequent production of A.M.D. These measures along with the water treatment processes and research programs will be outlined in the paper.

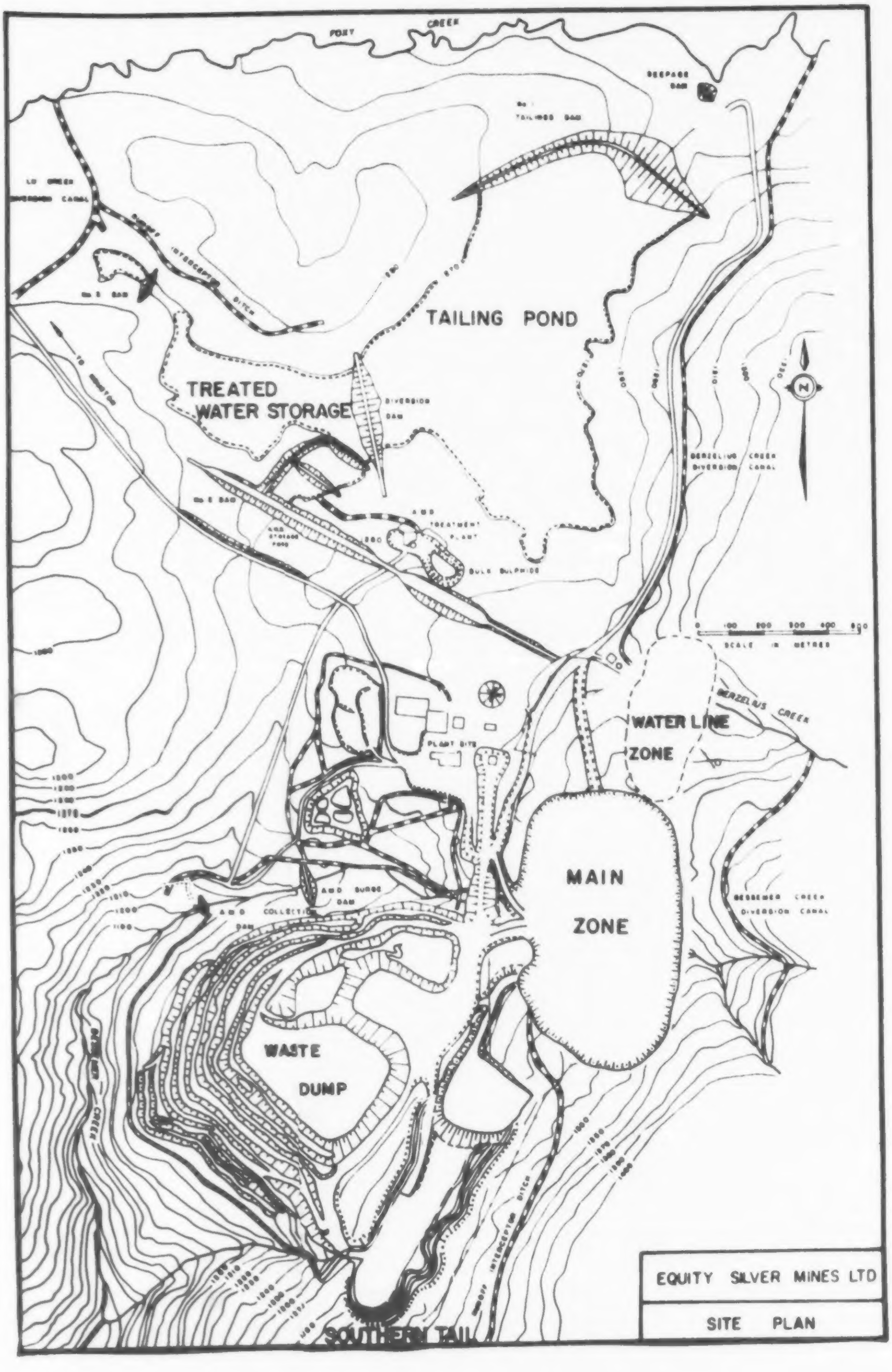
FOXY CREEK
SEEPAGE DAM
TAILINGS DAM
TAILING POND
TREATED WATER STORAGE
DIVERSION DAM
A.M.D. TREATMENT PLANT
BULK SULPHIDE
BERZELIUS CREEK DIVERSION CANAL
0 100 200 300 400 500
SCALE IN METRES
BERZELIUS CREEK
WATER LINE ZONE
PLANT SITE
MAIN ZONE
BESSEMER CREEK DIVERSION CANAL
A.M.D. SURGE DAM
WASTE DUMP
INTERCEPTOR DITCH
SOUTHERN TAIL
EQUITY SILVER MINES LTD
SITE PLAN

FIGURE 2: EQUITY SILVER MINES LTD. SITE PLAN

KINETICS OF ACID GENERATION

The degree to which acid generation occurs varies with the nature and concentration of available sulphide (pyrite) material. Pyrite mineralization at Equity Silver ranges from disseminations to the massive form. Where these massive forms occur along fracture planes, liberation and exposure due to blasting is thought to be maximized, likewise acid generation processes.

KINETICS OF ACID GENERATION

$$FeS_2 + \frac{7}{2} O_2 + H_2O \rightarrow Fe^{++} + 2\,SO_4^{=} + 2\,H^{+} \qquad (1)$$

$$Fe^{+++} + \frac{5}{2} H_2O + \frac{1}{2} O_2 \rightarrow Fe(OH)_3 + 2\,H^{+} \qquad (2)$$

$$Fe^{++} + \frac{1}{4} O_2 + H^{+} \rightarrow Fe^{+++} + \frac{1}{2} H_2O \qquad (3)$$

$$FeS_2 + 14\,Fe^{+++} + 8\,H_2O \rightarrow 15\,Fe^{++} + 2\,SO_4^{=} + 16\,H^{+} \qquad (4)$$

FIGURE 3: TYPICAL OXIDATION REACTION FOR SULPHIDE (PYRITE) MINERALS

Metal and acid forming ions liberated during the initial stages of oxidation, combine with alkaline materials inherent in the host rock and settle out of solution as soluble metal precipitates (Reactions 1 and 2 - Figure 3). With continued oxidation (Reaction 3) and the lowering of solution ph, metallic precipitates are taken back into solution. This step occurs when available alkalinity in the waste is depleted. Liberated metal ions, particularily iron in the ferric state (Fe+3), act as an oxidizer which further react with sulphide materials. As the process accelerates and the ph of the solution drops below 2.5, the rate of activity of the ferric iron (Fe+3) is governed by the combined effect of bacterial oxidation of ferrous iron (Fe+2), the reduction of ferric iron (Fe+3) by pyrite, (Reaction 4) and the associated formation of ferric sulphate and hydroxal complexes. If the reactions are allowed to proceed uncontrolled the cycle will continue until the sulphide supply is depleted.

ACID GENERATING POTENTIAL

In assessing the reactivity of an ore body, waste materials are tested for acid producing potential and buffering (alkalinity) capacity. Results of testing at Equity are quoted as a ratio, Acid Generating Potential : Acid Consumption Capability. For instance, an index ratio of 4.5, merely means testing has indicated that the waste material has the potential to produce 4.5 times more acid than it is capable of neutralizing. Where these ratios are low, kinetic tests in columns should be used to substantiate static acid-base accounting. Release of neutralizing capacity is greatly affected by factors such as particle size and hydroxide coating. Erroneous conclusions based on static test results alone are not uncommon.

The acid producing potential of pyritic waste material within the Main Zone Pit varies between a low of 0.15 to a high of 65, with the majority of values falling within the 10 to 20 range (Sperling, 1984). Zones of acid generating potential are illustrated in Figure 4.

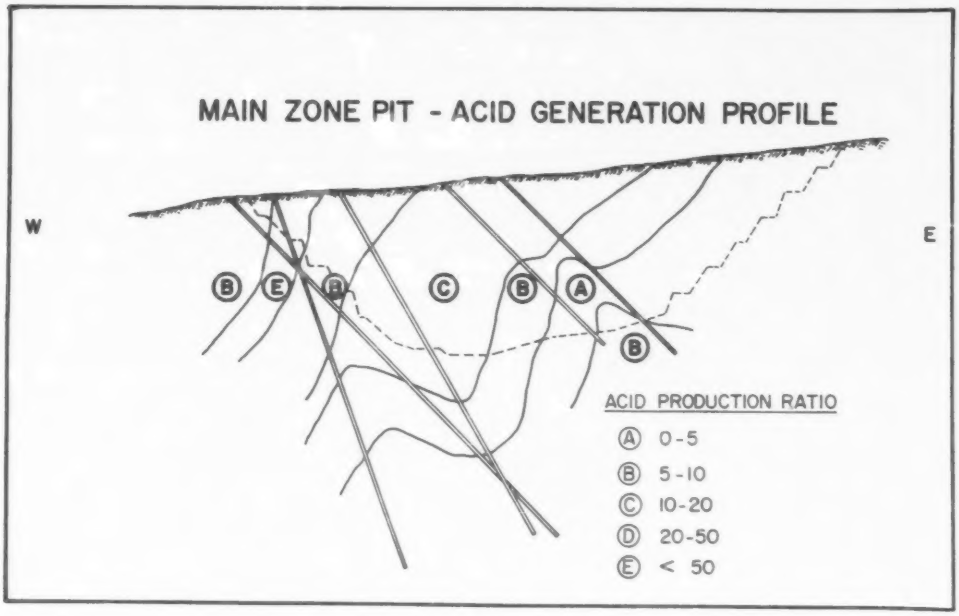

FIGURE 4: COMPOSITE CROSS SECTION OF MAIN ZONE ORE BODY WITH ACID PRODUCING POTENTIAL ZONING

Non-acid producing wastes occuring on the east wall of the Main Zone Pit are selectively mined for construction purposes and used at sites where A.M.D. must be eliminated. i.e. tailing dams. Use of non-acid producing materials for these projects precludes the possibility of reducing acid producing potential through acid-base accounting by strategic placement of wastes. All wastes classified as acid producers must be stockpiled on a common dump and reclaimed to reduce oxidation rates.

WATER TREATMENT AND SLUDGE HANDLING

Approximately 800,000 cubic metres of A.M.D. must be collected annually and processed through a lime neutralization treatment plant. Treated water from the plant complies with stringent quality objectives and is discharged to the receiving environment at dilution ratios governed by metal content and stream flow rates.

A.M.D. TREATMENT STATISTICS

VALUES IN mg/L

	ph	ACIDITY	SO_4	Cu(d)	Zn(d)	Fe(d)
RAW A.M.D.	2.35	10,000	8,500	120	80	800
TREATED	7.80	NIL	1,600	0.01	0.04	0.03

PUMPING COST	0.15 dollars / M^3
TREATMENT COST	0.70 dollars / M^3
ANNUAL VOLUME TREATED	800,000 M^3

Table 1 - A.M.D. Treatment Statistics

Estimated costs for pumping and treating A.M.D. at present acidity levels is in the order of $0.85 per cubic metre. Typical A.M.D. treatment statistics are illustrated in Table 1. The sediment (sludge) produced from the treatment process consists mainly of metal hydroxides. Yearly accumulation of this sludge represents not only a short term storage problem but a long term reclamation concern.

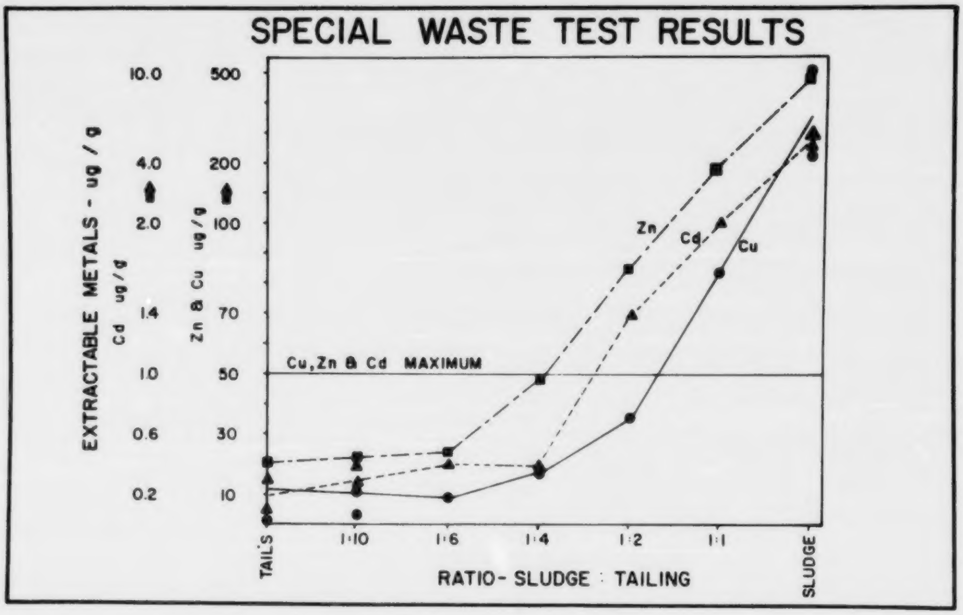

FIGURE 5 - RESULTS OF SPECIAL WASTE TESTING USING VARIOUS RATIOS OF A.M.D. SLUDGE AND TAILING

A.M.D. sludge, through testing, has been classified a Special Waste according to the proposed Special Waste Act. By mixing A.M.D. sludge with mill tailing discharge at ratios as low as 1:4, extracted metals from the composite meet guidelines laid out in the act. Figure 5 illustrates that tailing complies with Special Waste guidelines, whereas A.M.D. sludge far exceeds the limits set forth. Mixtures of A.M.D. sludge to tailing between the ratios of 1:10 and 1:6 show little change from baseline extractions within the tailing. Deviation from this trend becomes evident at ratios of 1:4 or less. Permitting has been completed to accommodate mixing the two products at a 1:10 ratio. Disposal of the material will be handled in this fashion during the period when the mine is in operation. Alternate options for sludge handling after abandonment include disposal in large ponds or beneath water within the Main zone pit.

The mechanics of pumping sludge has met with several operating difficulties. Pumping was initially carried out in a pond containing a surface layer of water. This resulted in a vortex around the pump with sludge being supported by water thus preventing lateral movement of the material. A waste rock divider dyke within the two cell arrangement was subsequently sealed with glacial till to prevent short-circuiting of water between the active pond and pond being excavated. Elimination of the water sources improved sludge pumping and at 5 percent solids sludge migrated over the full length of the pond supplying feed to the pump.

WASTE DUMP CONSTRUCTION

Disposal of all acid generating waste material is restricted to a common dump site (Figure 2). Prior to identifying A.M.D. occurrences, pyritic wastes were liberally used for fills throughout much of the property. Sites difficult to access for purpose of A.M.D. collection have required, where possible, excavation of acid generating material. Similar clean-up will be scheduled throughout the mine life to consolidate areas of contamination.

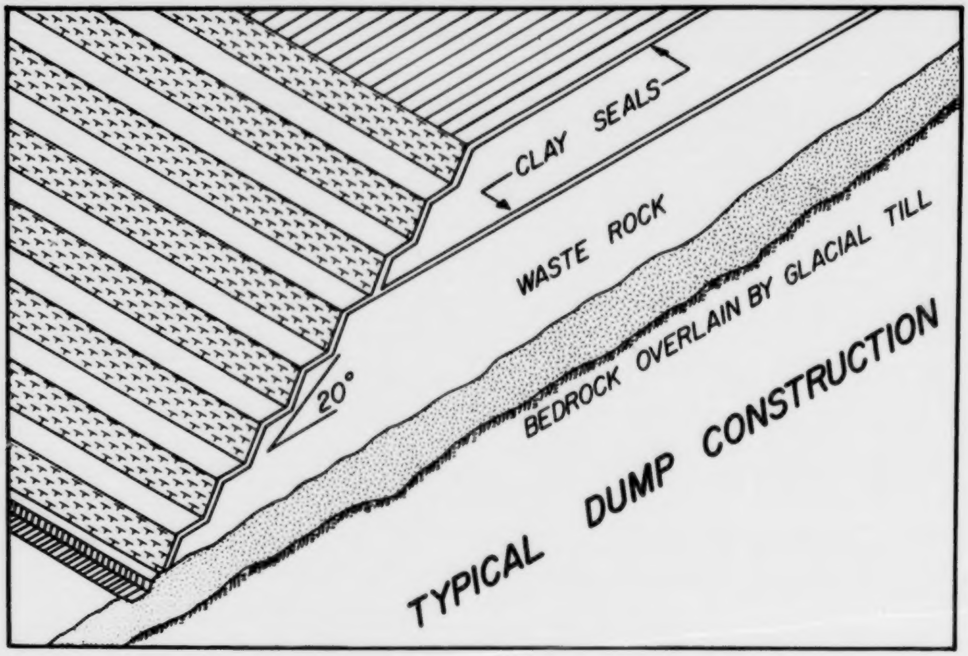
CLAY SEALS
WASTE ROCK
BEDROCK OVERLAIN BY GLACIAL TILL
20°
TYPICAL DUMP CONSTRUCTION

FIGURE 6: TYPICAL DUMP PROFILE AND IMPERVIOUS CLAY SEALS

Initially the waste dump was constructed by disposal along a common horizon. This construction technique produced high dump faces with static loads in excess of shear strengths of the underlying glacial till, consequently resulted in several minor localized failures. Geotechnical investigation (Klohn Leonoff, 1984) indicated that dump faces would have to be reduced from the natural angle of repose (37 degrees) to an overall slope of 20 degrees to achieve a safe weight distribution. Dump faces were reconstructed from the lower limits upwards, eventually merging with the existing dump head. In achieving the design slope angle, the dump had to be reconstructed in individual lifts of 10 meter thickness, each stepped in to conform to the 20 degree overall slope (Figure 6).

Reconstruction of the dump coincided with the period when A.M.D. occurrences became most prevalent. The new design accommodated a continuous reclamation program whereby completed berms and side slopes could be top-dressed with a one meter layer of glacial till and seeded to establish a protective vegetative cover.

Air exchange within dump structures accelerates during the winter months as gases produced from the oxidation process set up convection currents during periods of maximum atmospheric pressure variation, much the same process as in a fireplace chimney. To further reduce this air exchange, an intermediate layer of till was placed within the dump during the course of construction (Figure 6).

Glacial till is removed from active mining areas and from stockpiles inventoried during the pre-production stage. Disposal of till for reclamation purposes during the mining cycle precludes the need to stockpile material and in turn decreases final abandonment costs. Glacial till hauled to the dump represents a small incremental cost over and above normal operating expenditure.

To accommodate waste disposal demands, the dump will be extended easterly (Figure 2) to backfill the Southern Tail pit. By placing a portion of the waste below water within the pit, oxidation processes will be eliminated.

Wastes (especially the fine fraction) placed below the water table provide a supply of available alkalinity. Provided the acid generating processes within the area are minimized, residual alkalinity should maintain water quality within permit specification. A one meter layer of inert material (Gabbro) was placed at the appropriate horizon to accommodate fluctuation in the water table. This will safeguard alternate wetting and drying of pyritic wastes that would otherwise contribute to A.M.D. generation. A spillway has been constructed at the south end of the pit to ensure that a constant water level is maintained. Those materials placed above the water table will be encapsulated with a clay layer to decrease oxygen diffusion.

Results of existing reclamation efforts have yet to be fully assessed, although where these measures have been instituted at localized sites there has been a downward trend in acidity and dissolved metal concentrations. Monitoring programs are presently being developed or modified as assessment tools.

Temperature probes installed in the southern flank of the dump, have been used over the past 3 years to monitor internal dump temperatures. Observations to date reveal that the reaction produced through oxidation has elevated internal temperatures to 55 degrees celsius. A recent decline of several degrees has been observed and is thought to be a product of reclamation measures. A program of thermal imaging of pits and dumps has been recently completed by Environmental Protection Service based in Vancouver, B.C. . The practice of photo heat tracing (Thermography) is presently under investigation as a tool to assess the effect of till covers on pyritic wastes and a detection method for pin-pointing problematic zones yet to be reclaimed.

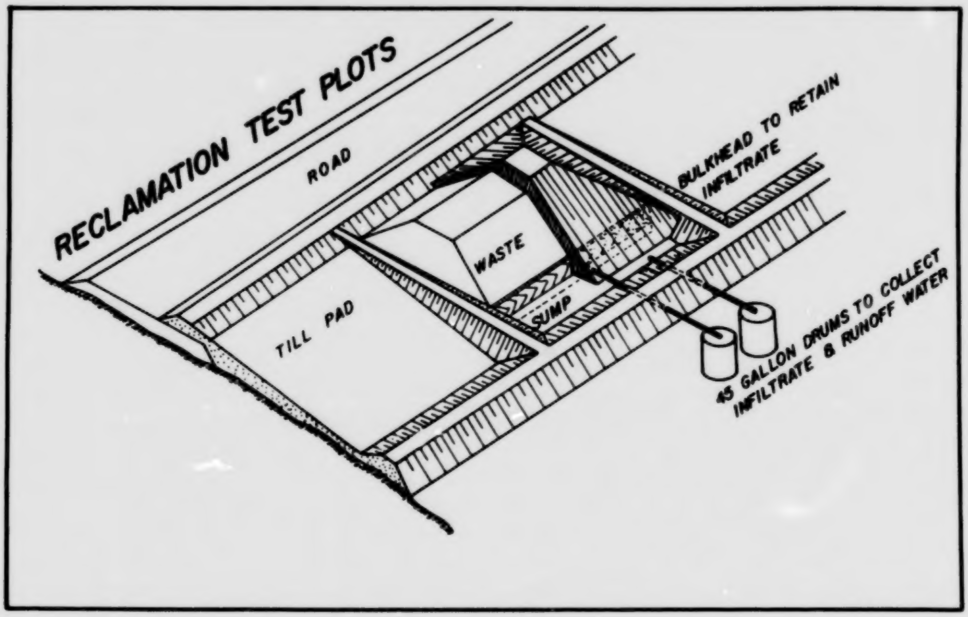
RECLAMATION TEST PLOTS
ROAD
TILL PAD
WASTE
SUMP
BULKHEAD TO RETAIN INFILTRATE
45 GALLON DRUMS TO COLLECT INFILTRATE & RUNOFF WATER

FIGURE 7: SCHEMATIC OF RECLAMATION TEST PLOTS

Test plots (Figure 7) representing scaled down versions of the waste dump have been developed to evaluate present reclamation measures. Glacial till covers have been added to wastes at thicknesses comparable to present use on the waste dump. Uncovered control units will be used to assess the effect of present till thicknesses.

A qualitative water monitoring program has been in place since 1983 to evaluate the impact of reclamation efforts. This program has recently been expanded to provide hydraulic balances of runoff. Hydraulic balances, along with metal analyses, will be used to calculate contaminant flows out of the waste dump and to delineate problematic zones.

WASTE ROCK AMENDMENTS

The anionic surfactant (Sodium Lauryl Sulphate) has been tested to evaluate bacterial suppression methods within waste fills. Although the initial lab testwork looked encouraging, results of in-field testing proved to be less than favourable. Nevertheless, use of this product should not be overlooked pending favourable conditions. To be effective, a sufficient dosage of the soap product must be available to completely infiltrate the spoil. Application rates at Equity Silver were based on treating the top meter of waste as outlined by past experience at coal operations. Recent findings at Equity Silver indicate that this oxidation zone, in coarse rock, can be in excess of 10 meters. Because of this oxidation depth, spoil piles could not be effectively saturated with treatment solutions. Recent tests carried out by the U.S. Bureau of Mines on Equity waste material support this assessment (Watzlaf, 1986). As a further complication the surfactant used biodegraded rapidly under the acidic conditions thus would require reapplication more frequently than the two to three months period experienced in the coal fields.

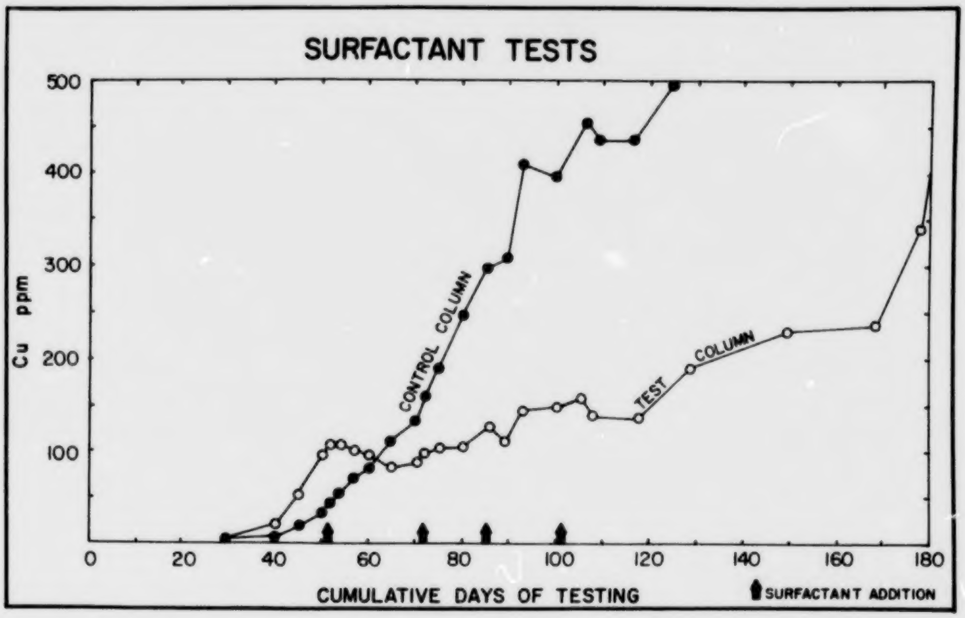

FIGURE 8: EFFECT OF S.L.S. ON OXIDATION RATES

Results of lab simulations using sodium lauryl sulphate (S.L.S.) are illustrated in Figure 8. Where S.L.S. was used in a test column containing pyritic waste, oxidation rates and dissolved metal content were suppressed in relationship to an uncontrolled test. Copper was used as a trend indicator, although other metals, sulphates or acidity values could have been similarily used. After surfactant addition was discontinued, copper values in the test column approached the trend set in the control column, thus indicating re-population of bacterial colonies and acceleration of oxidation processes.

Precipitates formed from the treatment of A.M.D. consist of metal hydroxides and residual lime. Injection of these precipitates into waste rock voids was initially considered an option for sealing and adding alkalinity to pyritic wastes. Tests were carried out to evaluate the impact of this amendment on oxidized pyritic waste rock.

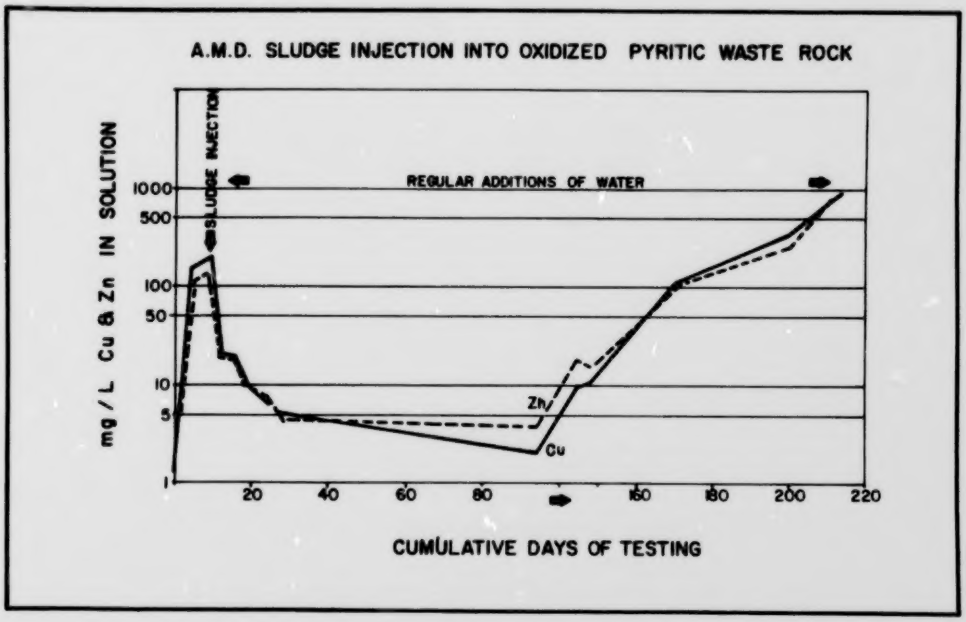

FIGURE 9: A.M.D. SLUDGE INJECTION INTO OXIDIZED PYRITIC WASTE

Injection of the product revealed a significant improvement on water quality. However, over an elapsed period acid generation processes within the waste exceeded the buffering capacity of the sludge resulting in a deterioration of effluent quality. Sludge injected into the waste was redissolved, further adding to the metal loading in the effluent (Figure 9). Under less severe conditions alkaline sludges may be a suitable amendment for control of A.M.D.. Operational aspects of a scaled up program were not investigated nor the geotechnical effect on dump structures.

SIMULATIONS

Disposal and abatement concepts are tested in columns containing the rock types in question. These test vessels, constructed from five foot lengths of P.V.C. pipe, are equipped with either porous or impervious bases depending on the water course regime being evaluated. Solutions are circulated through each column by purging air through plastic tubing connected to the effluent receptacle and top of column.

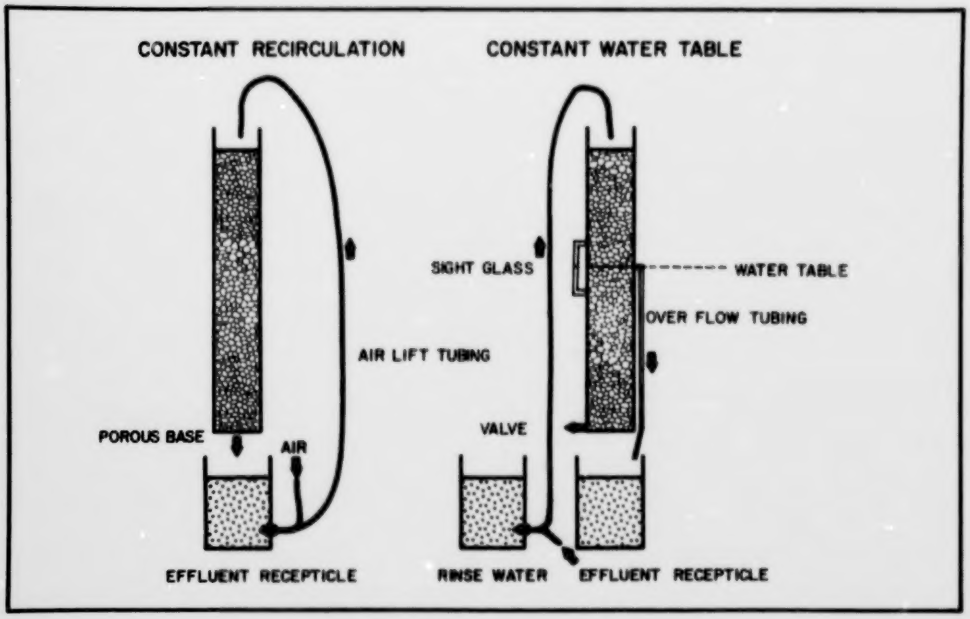
CONSTANT RECIRCULATION
CONSTANT WATER TABLE
SIGHT GLASS
WATER TABLE
OVER FLOW TUBING
AIR LIFT TUBING
POROUS BASE
AIR
VALVE
EFFLUENT RECEPTICLE
RINSE WATER
EFFLUENT RECEPTICLE

FIGURE 10: COLUMN TESTS

Acid generation processes can be accelerated by subjecting wastes to a constant circulating load of effluent (Figure 10). Essentially this technique enhances leaching by cumulating acidity and bacterial concentrations and works well for testing products such as surfactants as an abatement measure.

Similar columns, with sealed bases, have been used to evaluate oxidation processes below a fixed water table. Water is held within the lower section of the column by sealing off the base and providing a drain midway to decant off water introduced through the top section (Figure 10). The decant serves as a sample point for monitoring water quality in the upper horizon above the water table with substrate water quality being monitored from a valved under drain. Results from these tests indicate minimal oxidation of pyritic waste while in the submerged state.

RECLAMATION

Reclaimed areas are revegetated to stabilize glacial till covers and reduce precipitation infiltration through evapotranspiration losses. Various grass and legume species are evaluated using test plots for selection of suitable mixtures. Assessment of these plots is carried out annually and documented for future reference. Seeding programs employing hand and aerial methods have been reasonably effective in establishing vegetation on well graded and unconsolidated soil types and to a lesser degree on the dense consolidated glacial till, especially on side slopes. Hydroseeding methods were subsequently tested and found to provide better results. As results of the test work were encouraging, Equity Silver has since purchased a hydroseeder to supplement traditional seeding techniques.

Although reclamation efforts are primarily geared towards reducing oxidation within wastes dumps, the aesthetic and habitat benefits are also deemed important. An increase in deer population has been observed over the past few years and is thought to be a product of reclamation efforts.

CONCLUSION

Control measures at Equity Silver Mines Limited have been effective in minimizing the environmental impact of A.M.D.. Reclamation practices appear to have reduced oxidation rates at confined sites, although considerably more monitoring will be required to fully evaluate control measures on the larger disposal site. Alternate methods tested for suppressing acid generation processes initially appeared encouraging however have since been curtailed due to questionable long term benefit. With existing reserves in the neighbourhood of 5 years, plans are now being developed to prepare for mine abandonment. This will require a host of studies associated with sludge disposal, tailing pond abandonment and further research into A.M.D. mitigation methods for the waste dump.

ACKNOWLEDGEMENT

As spokesperson for Equity Silver Mines Limited, the author would like to thank the executive of the Acid Mine Drainage Workshop for the invitation to present this paper, and to the management and operating crews at Equity Silver for their undivided support in dealing with environmental concerns.

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THE IN-SITU MITIGATION OF ACIDIC DRAINAGES - MANAGEMENT OF HYDRO-GEOCHEMICAL FACTORS

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Abstract. Acidic drainages are closely associated with excavations of calcium carbonate deficient/pyrite enriched strata (overburden). During this process pyrite oxidizes to produce acidic waters containing large concentrations of iron and sulfate. In the low pH environment, iron bacteria catalyze the reactions, thereby accelerating the oxidation of iron and pyrite and creating large acid loads (Load = Concentration x Flow Rate). Mitigating procedures are available to control the rate of acid production. These include phosphate complexes, plastic liners, bactericides, selective overburden handling, and topical application of alkaline/calcareous materials. However, in most cases, the geochemical measures are used to reduce the acid concentrations, ignoring to a large extent the flow components of the drainage quality that could be potentially manipulated to increase the effectiveness of treatment.

An innovative concept, whereby surface water is recharged to a mine backfill and whose quality is controlled to affect substantial reductions in drainage acidity, is presented. The alkaline trench technology is designed to induce and facilitate infiltration to a porous backfill. The recharge is enhanced with alkalinity to displace the iron bacteria, neutralize the acidity, and reduce pyrite oxidation. Several field tests in central West Virginia demonstrate a 60-75% reduction in acidity during the first year following implementation.

STATEMENT OF THE PROBLEM

Acidic drainages form under natural conditions when carbonate deficient strata containing pyrite are exposed to the atmosphere and accelerated oxidizing conditions. During this process a variety of iron sulfides (FeS-FeS_2) oxidize in the presence of oxygen and water to form soluble hydrous iron sulfates that commonly appear as white and yellow salt crusts on the surface of weathered rock faces. In contact with water these compounds hydrolyze, forming acidic, sulfate and iron enriched drainages. Ferrous and ferric oxyhydroxides impart the red and yellow color characteristic of acid drainage and may precipitate to form the "yellow boy" that is commonly observed in the acid impacted streams (Caruccio and Geidel, 1978).

The chemical reactions explaining the oxidation of FeS_2 and the production of acidity (H^+) are given by the following equations:

$$2FeS_{2(s)} + 7O_2 + 2H_2O = 2Fe^{2+} + 4SO_4{}^{2-} + 4H^+ \tag{1}$$

$$Fe^{2+} + 1/4\ O_2 + H^+ = Fe^{3+} + 1/2H_2O \tag{2}$$

$$Fe^{3+} + 3H_2O = Fe(OH)_{3(s)} + 3H^+ \tag{3}$$

$$FeS_{2(s)} + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4{}^{2-} + 16H^+ \tag{4}$$

(Barnes and Romberger, 1968; Baker, 1975)

The stoichiometry of Eq. (1) shows that 1 mole of FeS_2 will produce 2 moles of H^+ (acidity). In turn, the Fe^{2+} generated by the reaction of Eq. (1) can readily oxidize to Fe^{3+} and produce additional H^+ (Eq. (3)). References cited by Baker (1975) show that FeS_2 can be oxidized at a more rapid rate in the presence of excess Fe^{3+} in solution (Eq. (4)).

The oxidation of Fe^{2+} to Fe^{3+} proceeds slowly under normal conditions, as in Eq. (2) and (3) (Singer and Stumm, 1970). However, several workers investigating the oxidation rates of sterilized versus inoculated samples showed that certain iron bacteria catalyze the oxidation reactions involved in acid formation and effectively increase the acidity produced (Kleinmann _et al._, 1981; Walsh and Mitchell, 1972).

These bacteria are viable in aqueous environments with pH values that range from 2.8 to 3.2; the viability decreasing with increasing pH. As such, the pH of the ground water controls the occurrence and distribution of these bacteria (Poissant and Caruccio, 1986).

TREATMENT MECHANISMS

Large-scale Neutralization. With large flows and high acid loads, large-scale, manufactured treatment facilities are used to neutralize the acidity, oxidize the iron, and generate a high-density settleable sludge. Many of these systems are available on a commercial basis.

Small-scale Chemical Neutralization. Because of the relatively simple and controlled aspects of dispensing a caustic medium, most mining operations neutralize acidic drainages in the vicinity of seep emanation. Acidic seeps are diverted into one flow channel which may be weired to monitor flows by a modified Parschall flume. A float in tandem with a valve dispenses sodium hydroxide (caustic) directly into the acidic flow, the dosage rate being determined empirically. The pressure head for the caustic is provided by locating the storage tanks at a higher elevation (and adjacent to roads to allow supply trucks easy access to refill). A narrow diameter (1-2 cm) PVC pipe conveys the caustic from the storage tank to the dispensing valve. The scheme does not require electrical power and is well suited for treating acidic waters in remote parts of the mine. The viscosity of the caustic, however, is readily affected by temperature changes and the dosage rates must be continually monitored and adjusted during the cold winter months. Further, during periods of heavy rainfall, runoff augments the flow into the monitoring device, which activates additional caustic despensing for flows that may have lower acid loads.

Combinations of sodium carbonate briquettes, calcium oxide, potassium hydroxide, and various oxidizers (peroxide, sodium-calcium hypochlorite) may also be used in the treatment. Each of these treatments produces a sludge that is retained in settling ponds. The ponds are cleaned and the sludge disposed of at the mine site.

Wetlands. Several studies have demonstrated the effectiveness of wetlands in treating the acidity, sulfate, and iron of mildly to moderately acidic mine drainages (Kleinmann et al., 1983). In these natural environments, sulfate is reduced to either bisulfide (with an attendant reduction in acidity) or pyrite (with the removal of iron) (Land and Wieder, 1982). Wetlands are created by impounding gently sloping low-lying areas and adding excessive amounts of organic matter (straw, hay, etc.). Following a brief period of decay, a sphagnum moss culture is introduced to the system.

The advantages of the treatment are the low maintenance and the creation of an aesthetically appealing treatment facility. However, the ecosystem providing the base of the treatment may occupy only certain natural niches. Some researchers, however, have developed and selectively bred several moss species that may be used across a variety of conditions. Some question exists as to the levels of acidity and iron that the natural system may accommodate before being detrimentally affected by the acidic waters.

CONTROL MECHANISMS

The natural acid-producing elements are pyrite, oxygen, water (moisture), and the catalytic effects of bacteria and water geochemistry. All but the pyrite, which comprises an intrinsic part of the rock matrix, can be directly impacted by in-situ abatement techniques to reduce acid concentrations, flow rates, and acid loads.

Detergents and Bactericidal Additives. Limiting the oxidation of ferrous to ferric irons effectively reduces the amount of acidity produced by almost half. Restricting the acid reactions to the steps described by Equations (1) through (3) may provide acid levels that can be accommodated by the normal levels of bicarbonate alkalinity derived from carbonate dissolution. Should the reaction expressed by Equation (4) begin, the rate of acid production is greatly enhanced and the high concentrations of acidity will overwhelm the low solubility-limited calcareous-generated alkalinity. The addition of a suitable bactericide limits the rate of acid production and may be used to control the acid loads (Kleinmann and Erickson, 1983).

This treatment measure can be applied to active as well as abandoned mines. When used to complement other treatments, it is an effective control measure and because the bactericide is water soluble and entrained by the recharge events, it can penetrate deep within the backfill or mine portals and affect remote sections of the mine space. Contingent upon the flow through rate/time, it may be necessary to apply the bactericide frequently.

Phospate Complexes. Phosphate compounds (primarily in the form of pulverized apatite) introduced to an acid system complex the free iron radical and inhibit the hydrolysis reaction, thereby reducing the amount of acidity produced (Flyn, 1969). The phosphate is solubilized by only slightly acidic conditions and consequently must be physically juxtaposed to the pyrite reaction site. Some field tests using phosphate mine slurry have provided varying degrees of success and some failures have been linked to a mass transfer problem. Accordingly, the application of phosphate is best completed with the placement of the backfill where admixture with the acid spoil material is possible (Meek, 1984).

Because it is an iron complexing technique, acid sources associated with iron hydrolysis will be eliminated. The levels of acidity associated with the reaction (Eq. (1)), however, will continue to be produced.

Selective Placement of Toxic Material. Obviously, this strategy is incorporated as part of the mine plan and implemented during the backfilling reclamation phase. The West Virginia Acid Mine Drainage Task Force Recommendations advocate, in part, the placement of toxic spoil on top of a highly permeable inert material pad about 3-5 m thick. In turn, the acid material is capped with a clay seal, covered with 1 m of soil, and the surface is reclaimed. In a field test, a small (6 ha) hydrologically-isolated hill was reclaimed according to these recommendations and monitored through time. The results showed the clay seal to effectively prevent infiltrating waters from contacting the toxic spoil, and that the porous pad was instrumental in keeping the water table below and out of the influence of the acid material. Through a combination of bromide and iodine tracers and oxygen isotope data monitored over a 2-yr period, this method was demonstrated to be effective in preventing acid drainage formation by the toxic spoil material. However, this technique did not eliminate the acidity produced by the pavement (Geidel and Caruccio, 1984).

Plastic Liner. Reducing the flow of water through acid material, by reducing the infiltration capacity, effectively reduces the acid load of the system. In one technique the surface of the acid producing mine is graded and covered with a 20 or 34 mil continuously sealed plastic liner (not unlike those commonly used in hazardous waste sites). To protect the plastic from ultraviolet ray exposure, which decomposes the plastic, a 1 m layer of soil is used as a cover and vegetated for mass stability.

In a field test involving a 16 ha mine site where a 20 mil plastic liner was used, the net acid load emanating from two highly acidic seeps was reduced to a level sufficient to amortize the cost of the plastic liner within a 6-yr period (Caruccio and Geidel, 1983).

For this technique to be totally effective, however, the plastic liner should extend to the base of the backfill. The outslopes of most spoil banks are generally too steep to accommodate machinery easily or to maintain slope stability necessary for soil retention. Accordingly, a significant portion of the mine is left exposed (unlined). Avenues for recharge then exist and under some climatic conditions, the head developed at the outslope is sufficient to drive the recharge-wetting front under the plastic (temporarily reversing the hydraulic gradient). Subsequently, acid leachates drain from beneath the plastic when the normal gradient is reestablished.

This technique is labor intensive and lends itself best to active operations. If the acid material could be identified, concentrated in one area, and encapsulated in plastic, this procedure may be used in reclaiming abandoned mines.

Blanket (Surface) Application of Limestone. Acid producing reactions of acidic systems are sufficiently mitigated by alkaline recharge to warrant consideration of the topical use of calcareous material. One method of altering the mine water chemistry is through the application of a blanket veneer of limestone on the surface of the mine. Rainwater or snowmelt, in contact with the limestone, creates an alkaline wetting front. Under these conditions, the elevated pH displaces the bacteria, and pyrite oxidation is inhibited. Because of the limited solubility of calcareous material, however, the amount of bicarbonate produced is lower than the acid generated by pyrite oxidation. As a result, the initial alkaline wetting front has a negligible effect on the total acid system, and the mitigation of acid flow is not recognized until several recharge events, spanning as much as 6 or 8 years, have occurred (Geidel and Caruccio, 1982).

This technique is best suited for mines with highly permeable surfaces and mildly acidic drainages. In areas where the slopes are too steep for normal reclamation methods, helicopters may be used to dispense the limestone on inaccessible areas while four-wheel drive lime spreaders are used throughout the remainder of the mine.

Induced Alkaline Recharge Zones. The blanket application of limestone has two disadvantages that diminish the treatment's effectiveness. First, the ameliorant is not concentrated in the major recharge areas. As a result, the major flow paths and recharge avenues do not contact much of the limestone. Second, the limited solubility of the limestone produces low levels of alkalinity which do not significantly impact the more soluble acidic systems (Geidel, 1979).

A treatment strategy designed to overcome these two deficiencies (the hydrologic uncertainty and the low alkalinity produced by the limestone) forms the basis of the induced alkaline recharge technique. Here, major surface water flows are intercepted by shallow (1 m) trenches that are lined with a layer of sodium bicarbonate briquettes (applied at a rate of 1 kg per 0.5 m) and covered with 0.3 m of agriculture limestone reject material. In this process surface runoff is converted into highly alkaline (the effect from the briquettes), high flow (hence large alkaline loads) recharge events that are capable of overwhelming the acidic systems. Catalyzing bacteria are displaced, the acid system neutralized, and the pyrite oxidation stabilized. The strongly alkaline system lasts only until the briquettes are completely dissolved. Normally, this takes about 6 months, during which time the acidic system is impacted by highly alkaline waters. Once achieved, the limestone serves to provide sufficient alkalinity to maintain the non-acidic state (Caruccio and Geidel, 1984).

This technique may be easily applied to abandoned mined lands with a minimum amount of earth-moving effort. The trenches must be located in permeable zones to preclude trench flooding that may contaminate adjacent areas with sodium and have a deleterious effects on the vegetation.

Our field studies have shown that, in general, ground water flow through a backfill is highly channelized and that acidity is formed by the leaching of toxic rocks intercepted by these flow paths. By deliberately channeling water into constructed recharge zones, which are loaded with alkaline material, the induced alkaline flows form a greater percentage of the hydrology of the mine and should affect the seep quality accordingly.

The alkaline trench concept has been implemented and tested at an inactive surface coal mine in central West Virginia. The historical development leading to the emplacement of these trenches begins in January, 1982. At this time, five depressions on the surface of the mine were located and flagged for treatment. We thought the depressions to be major ground water recharge zones and directed the placement of alkaline material in these zones to affect the mine drainage quality. During the early part of February, 1982, the depressions, which ranged from one to three ha. in size, were treated with 25 to 110 kg of sodium carbonate briquettes and 13 to 166 tons of limestone reject. Each was spiked with a bromide tracer.

Samples from seeps were collected routinely and measured for temperature, pH, acidity, alkalinity, specific conductance, sulfate, and the tracer.

At the time that the depressions were treated, the ground was frozen, yet the seeps continued to flow and remained acid. During March - April, 1982 (the spring thaw), a sharp bromide peak was reported in one of the seeps with an accompanying decrease in acidity (dropping from 650 mg/l to 150 mg/l), sulfate (from 725 mg/l to 250 mg/l) and associated metals. In retrospect, because the tracer appeared in the seep at the time of the spring thaw, we now believe that the flow of water from the treatment to the seep was near surface interflow and did not penetrate the backfill.

For the remainder of the year, through December, 1982, traces of bromide were detected (though not at the levels measured during the spring) and the seep acidity returned to the levels before treatment.

During the summer of 1983, plans were made to recontour the site for the purpose of diverting water away from the highwall and affect runoff to the outslope. With a 30 foot backhoe, 16 large diameter holes were dug to pavement to collect samples for analysis and determine the configuration of the water table in the backfill. To our surprise, only 3 of the 16 holes encountered water. We further discovered that the zones below the depressions were generally dry. Thus, what we had earlier identified to be recharge areas, were, in fact, areas with perched water tables.

Further, in one hole dug to pavement (No. 13), water was not encountered until the backhoe had excavated 23 feet into the backfill. Within one hour, the water level in the hole rose to within eight feet of the mine surface. This suggests that the mine backfill is comprised of highly permeable channels that are randomly located and interconnected to create a "pseudo-Karst" hydrologic setting (Figure 1).

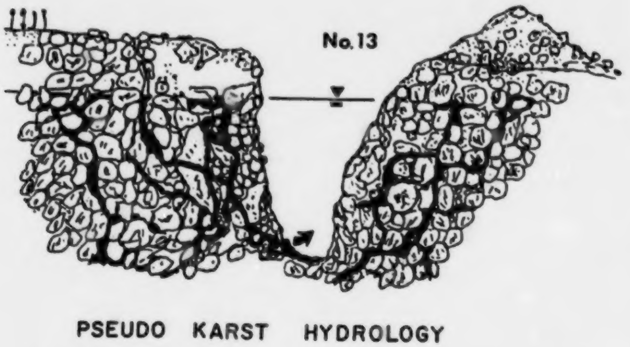

Figure 1 - General Representation of the Hydrologic Character of the Backfill

With this concept, the alkaline trench technique gains considerable support as a viable in-situ treatment of acid drainage. Compare, for example, the general broadcast of an ameliorant over the entire mine surface through which a small portion of the total treatment is intercepted by infiltrating waters and entrained to the backfill. In contrast, by concentrating the ameliorant along the major recharge zones, large effects are realized.

During active rain events, we often observed recharge taking place along the base of the highwall due to the slope of the backfill and the bouldery nature of the backfill (Figure 2). We believed that this zone could be loaded with alkalinity which would significantly impact seep quality. We recommended the emplacement of an alkaline trench along this zone and, in further development, the alkaline trench concept

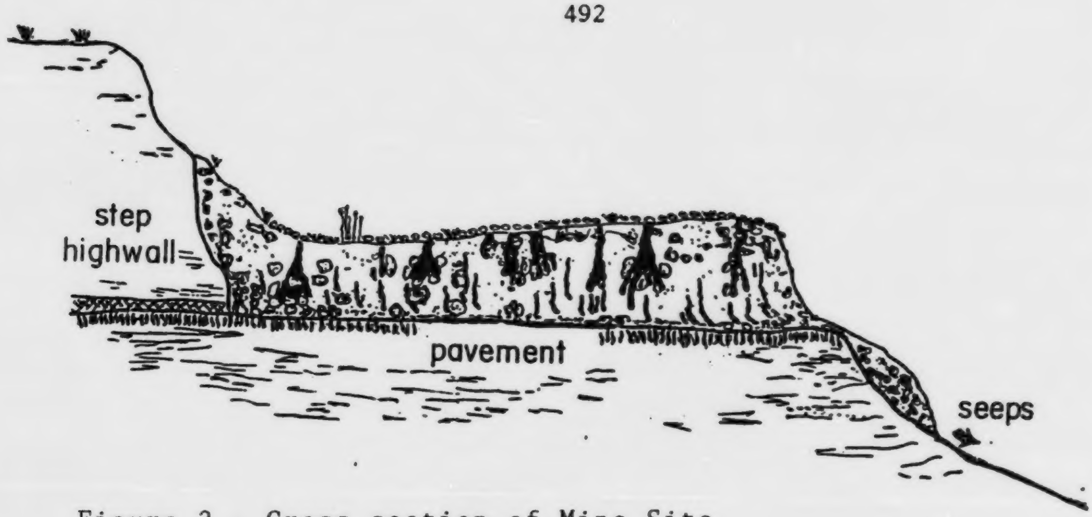
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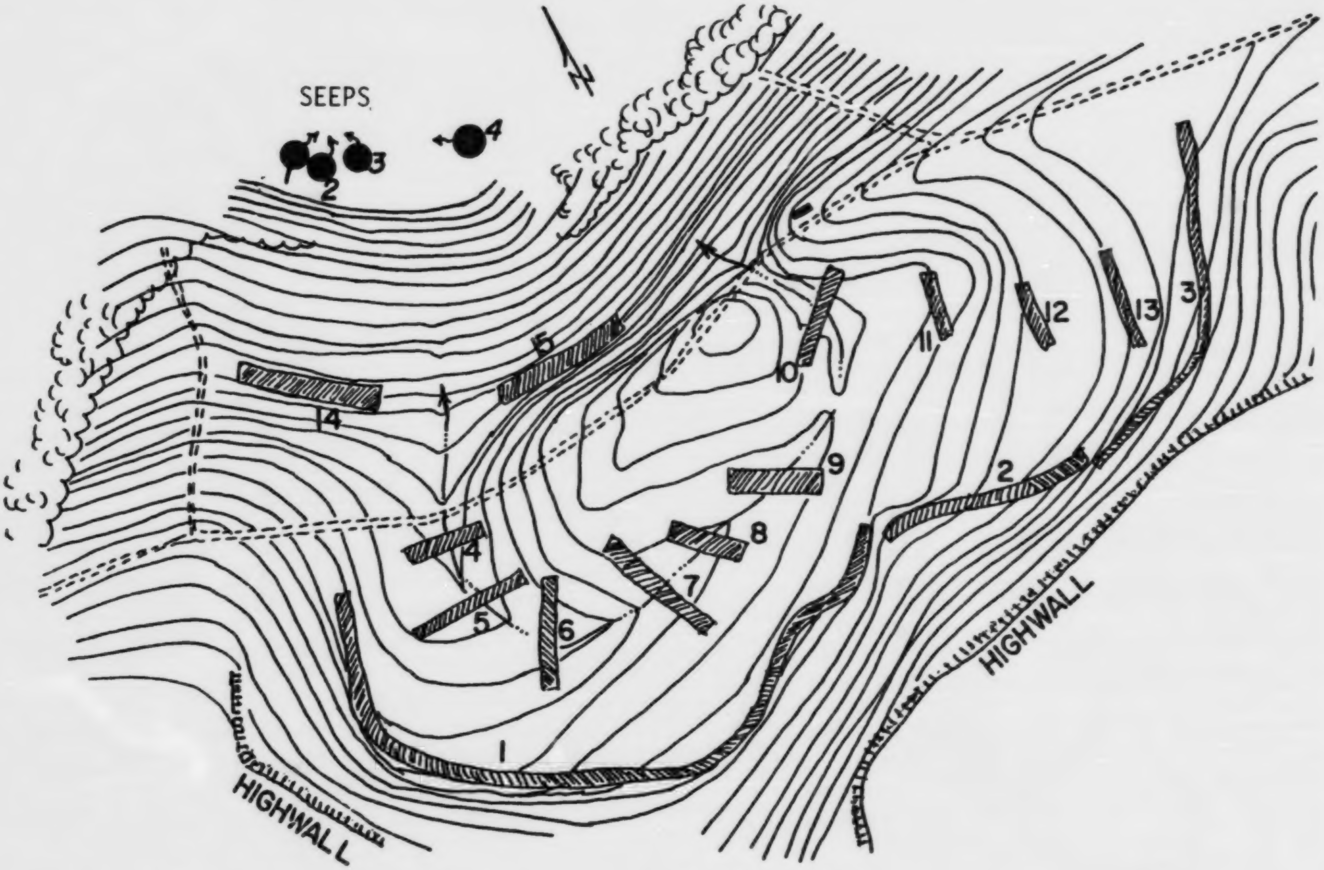
Figure 2 - Cross section of Mine Site

was fully implemented at the site. Figure 3 shows the configuration and locations of the alkaline trenches.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HIGHWALL

Figure 3 - Overview of Trench Construction and Locations of Seeps

Using a small bulldozer, a series of trenches averaging 3 m wide, 1 m deep and 20 to 215 m long were dug into the surface of the mine. Figure 3 shows the topography of the site and the locations of the trenches and seeps. The floor of each trench was lined with approximately 2 kg of sodium carbonate briquettes per 1 square meter and covered with 1 m of limestone reject material.

During the time that the holes were dug with the backhoe and the water exposed, 500 grams of KI were added directly to the water table in Hole 7 and 500 grams of KBr in Hole 13. Unlike the previous placement of the tracers in the near surface horizons, these tracers were placed directly into the ground water regime and provide good indication of the ground water flow and emergence of the recharge event at the seep.

The emplacement of the trenches and treatment was completed during the summer of 1983 and since that time, Seeps 1-4 (Figure 3) have been routinely monitored for pH, specific conductance, acidity, alkalinity, sulfate, calcium, magnesium, iron, manganese, iodine, bromide, and selected trace metals. Preliminary indications of sodium are indicated by the brilliant yellow color imposed to the flame during AA analyses.

The results of these tests are summarized in Figure 4. Until August, 1983, Seeps 1-3 show a continued and steady increase in acidity and sulfate. Beginning in September, a large influx of iodine is detected which coincides with dramatic decreases in acidity and sulfate. If the primary

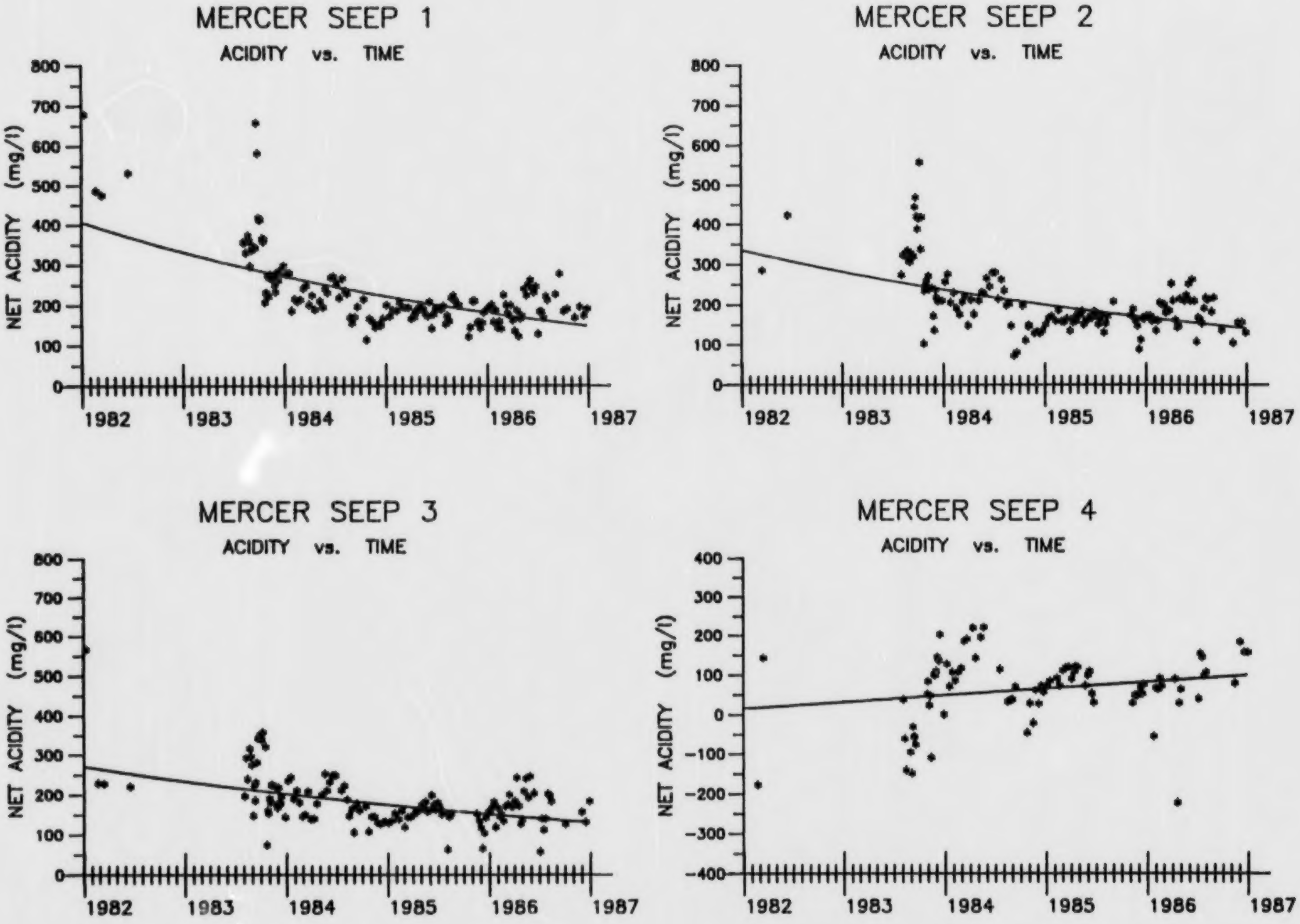

Figure 4 - Acidity trends versus time for seeps 1-4
(trench construction completed during Summer, 1983)

effect of the trenches were neutralization, the decline in sulfate would not have paralleled the decline in acidity. Sulfate is derived primarily from the oxidation of pyrite and the fact that sulfate is decreasing indicates the trenches are affecting the chemistry of the acid producing reactions.

In addition, the tracer appeared at the seeps approximately three to four months following its introduction into the water table. The linear distance between the point of introduction of the tracer (Hole 7) and the area of the seeps is approximately 500 feet. These data suggest that the ground water velocity within the spoil is on the order of 4 to 5 feet per day. Seep 4 originates as an alkaline seep at the time of the test due to the briquettes that were added directly to the hole during the excavation phase.

SUMMARY

Many treatment and control measures are available to mitigate and abate acid mine drainage from surface mines. The technique deemed most appropriate for a particular mine site depends on the age of the mine site, the degree of vegetation, the degree of calcareous material present in the overburden, the availability of a chosen ameliorant, accessibility of the site by equipment, and the source (physical and hydrologic) of the acidic seeps in the backfill or mine regime. These are but a few considerations to be examined in choosing an effective treatment technique. The site must be explored by drilling, the hydro-geochemical system understood, and the acid source identified before the treatment technology is implemented.

The encouraging results of the alkaline trench technique suggest that further study and consideration be given to the design of "leaky" mines in which the hydrology of the mine system be manipulated to provide increased alkaline flows and decreased acid flows to shift high acid load regimes toward alkaline ones.

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FACTORS TO BE CONSIDERED
IN
THE USE OF WETLANDS
TO TREAT ACID MINE DRAINAGE

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A PROBLEM associated with coal mining, ore mining, and clay mining is degraded waters. The water degradation takes various forms, but a great percentage (over 80%) in the Appalachian Coal Fields is associated with pyrite, and high sulfur coal (5% sulphur or more).

The problem with pyrites usually consists of low pH, high iron, and in certain geological strata, high manganese and aluminum concentrations.

The pyrite problem usually ends up with hydrogen ion concentrations (pH) varying from 0.9 to 6.9. This extreme latitude has been found within one county in the Pennsylvania bituminous coal fields.

The cost of upgrading this water to discharge standards exceeds one million American dollars per day in one state.

It is not unusual to find iron in excess of 2500 Mg/L, manganese in excess of 250 Mg/L, and aluminum in excess of 200 Mg/L in the same discharge. This author has found on several occasions high mineral loadings in the reclaimed, revegetated surface mines due to underground impoundments. These impoundments act as acid-generator stations, and eventually contaminate a very large area.

It behooves the mining operator to make certain while the mining operation is in progress that the pavement (or floor) of the operation has provisions for drainage. If the system can have free drainage, the possibility of a degraded water system is reduced.

It has been demonstrated on several occasions that by cutting through a low wall, or "digging out a seep," the underground impoundment is drained and the water quality vastly improves. An example is to have an iron concentration go from 400 Mg/L to 100 Mg/L, and on another occasion, to have a manganese concentration go from 525 Mg/L to less than 200 Mg/L. This shows that the less pyrite that is in contact with the water, the less acidic leaching takes place.

If the water has become degraded, it is still possible to manage the water and utilize biological systems to remove certain ions.

Biological Systems — Preliminary Requirements

To obtain the full benefit of a biological system, the water must be managed. This management consists of controlling water level, flow and velocity.

It is considerably more cost effective to manage the water, and prevent the conditions that cause deterioration, than to treat the degraded waters afterwards. Another consideration on water management is to clean up the pavement (or floor) of the mining operation, and not permit the water to degrade by being in contact with the waste material. Good housekeeping prevents many waters from deteriorating.

Consideration must also be given to diffuse flow. Diffuse flows are necessary to maximize the contact time to permit the biological systems to function. There is no question that the microsystem of the biological system does all the work of removing metal ions in the winter season.

In considering a biological system for degraded waters, the following parameters should be known:

1) Water flow, including seasonal variations
2) Mineral loadings, if present
    a) of iron (% ferrous/ferric ions)
    b) of manganese
    c) of aluminum
    d) of arsenic

    e) of selenium
    f) of lead
    g) of uranium

3) Hydrogen ion concentration
4) Acidity
5) Alkalinity
6) Total dissolved solids (projects precipitate buildup)

Besides various parameters for the water, climatic conditions are also big factors, and the following should be considered simultaneously.

1) Date of the last killing frost
2) Date of the first killing frost
3) Precipitation
    a) as rainfall (per month)
    b) as snowfall (per month)
4) Surge flows from precipitation
    a) snow melt
    b) 24 hour/10-year storm
    c) 24 hour/50-year storm

worst case conditions

Once the above parameters are known, consideration then can be given to biological systems.

Biological System Selection Consideration

The biological systems are climatic-zone specific; a water hyacinth system in Louisiana will not survive in Pennsylvania, and likewise, a typha/algae system may not survive in the high Northern Rockies.

Very often, it is necessary to find an endemic system that may be adapted for use. On many occasions, this author has found that, by studying local conditions, the biosystem that was endemic became the source of cultivars for man-made biosystems (with over 90% success).

Quite frequently, one can take a volunteer biosystem (and enhance this volunteer by proper water management), and induce the volunteer system to remove greater amounts of the metal ion than it had been able to do previously. This author has used this procedure, and it has worked on several occasions at pHs ranging from 3.5 to 6.5, and mineral loadings up to 250 Mg/L iron. The enhancement cost has averaged less than $0.25 (USA) per square foot, and the discharge meets NPDES standards.

The man-made biosystems are almost always site specific, but this author has found the following generalities do apply.

1) Hydrogen ion adjustment. This author has found that it is possible to remove 100 Mg/L of pyritic iron from solution, and still have a pH less than 3.0. It is necessary to instill a substrate capable of elevating the pH. The most economical material is a calcerous one, in the form of limestone or lime. For an elevation of more than two pH units, the material should have 100% available calcium carbonate equivalent. This material has operated over 2 years in raising water hydrogen ion concentrations from pH 2.8 to 7.2 on a continuous basis.

2) Organic strata. The beneficial microbiological system is now generally accepted as being heterotrophic. The organic strata that has been most successful is a hot composted system (used as a growing medium for the organisms), and serves the following functions, as it provides:

a) an almost sterile growing media (160 degrees F.);
b) a readily-available medium that can be produced anywhere from various materials;
c) an easily-handled material;
d) a non-competitive growing media without any inherent toxic material;
e) a media that can be buffered, and have nutrients added;

f) material with excellent percolating, and water-holding properties.

The minimum organic mass to be considered is six cubic yards (USA) per flowing gallon (USA)/minute. The minimum surface area exposed to the atmoshpere is 200 square feet per flowing gallon. All of the aforementioned criteria are for pHs ranging between 4.0 and 7.0, and for iron loadings under 150 Mg/L.

For hydrogen ion concentrations below pH 4.0, and iron loadings over 200 Mg/L, the above criteria require experienced water management. The design of the biosystem requires emphasis on cultivar selection, and feeding schedules.

The macro system requires the following considerations, as it is an integral part of the biosystem. The macro system should be:

a) climatic tolerant;
b) pH tolerant;
c) mineral-loading tolerant (wide range — 25 Mg/L to 1000 Mg/L iron);
d) reproductive under the worst-case conditions;
e) locally available, or readily obtainable.

Obviously, no single cultivar will fill the above requirements, and that these are both climatic-zone, and water quality specific. Such cultivars can be divided into four taxonomic groups.

1) Bacteria. The bacteria that are most desirable from a mineral-removing standpoint are the heterotrophic metal-ion oxidizers. The late R. W. Stone had found over seven species of bacteria that had an effect on removing manganese in the same humic strata sample. In an unpublished report R.W. Stone and B.G. Pesavento, present a graph showing a relationship between iron, manganese, and bacteria concentration.

2) Algae. The algae appear to have a multi-faceted tolerance as wide as bacteria. A table in a report, by L.R. Stark and B.G. Pesavento shows the tolerance of several species from various states in the USA.

3) Bryophytes. The bryophytes utilized are usually very site specific, and they are usually transplanted with one of the other cultivars. This author has had difficulty transplanting straight bryophyte systems — although bryophytes have been successfully used when they are utilized as secondary plantings with typha and sedges. Care in selection of bryophytes is required for successful transplanting, and water management is an absolute requirement.

4) Vascular plants. This author has been studying various forms of wetlands since 1950, and has been following over 40 volunteer units for the past 16 years. The following observations based on this experience are being passed on to the reader.

a) Typha (cat-tails)

This plant, on first appearance, has a very wide pH tolerance; the water flowing over its growing strata ranges from pH 2.0 to 10.0. However, the root system does not have this wide pH tolerance. Random sampling of root-system strata of volunteer wetlands (over several years) suggests that typha root systems can tolerate an actual pH range of 3.5 to 7.5. The substrates with higher pHs are plants that exceed seven feet in height, while lower pH substrates have plants only attaining 3-foot heights, including some showing diminished seed heads at heights of two feet. Such examples show the plants under great stress.

b) Sedges, Phragmites, Equisetum

The sedges and their relatives are often found intermixed in the ecological system with the typha. Pure sedge stands are more generally found in highland bogs at 2500 to 3000 feet above sea level, and greater heights in the Appalachian mountains. Sedges have been used,

> but this author has found that water management is critical with these plants — as they do not have the wide pH tolerance, or the water-level tolerance of the typha. However, it is not uncommon to see a clump of sedges growing in a totally decimated area. The spreading rate, and the transplant survival rate generally do not make sedges a prime candidate for selection as a vascular plant (especially if there is a more appropriate alternative). The phragmites and equisetum often volunteer into wetlands with degraded waters, but they are not generally used because of their narrow range of survival in decimated areas.

The primary consideration in designing a system is to provide a biological system that is self-regenerating, and also has the ability to absorb wide fluctuations of climatic conditions, water qualities, and flows. Such non-attended, self-regulating systems are quite complex, and efforts to explain many unanswered questions about these systems continue.

THE EFFECTIVENESS OF VEGETATION IN CONTROLLING ACID GENERATION IN BASE METAL TAILINGS

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ABSTRACT

Noranda Inc., which currently owns 18 mines in Canada and the United States, maintains a high degree of commitment to environmental protection. Reclamation research in the early 1970's at Noranda Waite Amulet site led to a successful vegetation program. However, the expected benefits of vegetation in reducing acid generation at Waite Amulet have not been realized.

Further research efforts have been undertaken at Heath Steele Mines Ltd., Newcastle, N.B., Waite Amulet, Rouyn-Noranda, P.Q., and a high carbonate tailings site in Ontario. The program involves a characterization of the geochemistry and hydrogeology and an evaluation of the parameters which control oxidation. Modelling tools will be evaluated for predicting the geochemical behaviour of these tailings with time and under varying reclamation scenarios.

Direct vegetation of reactive tailings has not restricted oxygen diffusion nor has it limited water infiltration. Consequently, oxidation of sulphide tailings is occuring, acid water containing heavy metals is being generated and complex geochemical processes of neutralization, precipitation, co-precipitation and adsorption are taking place to ameliorate pore water quality. The upper 1 to 2 m zone of oxidation is influenced by such factors as the water table and the sulphide content and particle sizing of the tailings. Infiltrating water due to precipitation and snowmelt transports pore water containing dissolved metals through the tailings and into the groundwater and surface seepage systems.

1.0 Introduction

Environmental protection is an important element of Noranda Inc's operations. In 1986, the Board of Directors approved an environmental policy which states the following:

> Noranda Inc. operations will strive to be exemplary leaders in environmental management by minimizing the environmental impact on the public, employees, customers and property, limited only by technological and economic viability. The following principles are basic to achieving this environmental objective:
>
> 1. The potential risks of new projects to employees, the public and the environment must be assessed so that effective control measures can be foreseen and taken, and all parties made aware of these facts.
>
> 2. Noranda group operations will implement site specific environmental, health, hygiene, safety and emergency response policies in the spirit of guidelines issued by Noranda Inc. as well as in conformation to applicable laws and regulations.

3. Noranda group operations will constantly evaluate and manage risks to human health, the environment and physical property.

4. Noranda group operations will be subject to periodic environmental, health, hygiene, safety and emergency preparedness audits.

5. A report on environment, health and hygiene, safety and emergency preparedness will be presented annually to the Board.

Though the policy is recent, the corporation has maintained a high degree of commitment to environmental protection over the years.

Noranda Minerals Inc., one of four Divisions within the corporation, currently operates, maintains or is decommissioning 18 mines in Canada and the United States. Of these, 10 are operating, 3 are being maintained in a closure mode awaiting improved metal prices or sale, and 5 are closed and are either being decommissioned or being prepared for sale. Among operating mines, 6 may enter the reclamation phase within 2-5 years unless additional reserves are identified through continuing exploration. Noranda Inc. tailings areas total approximately 3500 ha, of which 50% is classified as reactive due to the presence of sulphides.

Mining activities which may have a significant impact on the design of a reclamation plan - vegetation and acid generation being two of the parameters here-include:

- open pit operations in which case rock dumps may be formed - the rock representing the capping/stripping and boundary material surrounding the mineralized zone which must be removed to expose the ore, and
- development of a tailings pond which confines the remaining gangue material after the minerals are extracted from the ore.

Tailings and mine rock may be reactive, acid generating sources - the balance determined by the acid generating and consuming capacity of the host material. Today, techniques exist which can determine a positive or negative potential for acid generation. These technologies were not available when some of the early reclamation work was undertaken.

2.0 The Early Waite Amulet Vegetation Experience

It was not many years ago when a vegetative cover over a tailings area or a rock dump was considered to be the panacea. Noranda's history in this approach began in the

1960's when the current Horne Division initiated work on the Waite Amulet tailings pond near Rouyn-Noranda, Quebec. The objectives of these early efforts were to:

- control fugitive dust due to wind erosion of the surface;
- control surface water erosion;
- improve site aesthetics;
- improve the water balance by an active vegetative cover which would intercept, evaporate and evapotranspirate moisture thereby preventing its entry into and flow through the tailings mass;
- control acid generation.

The 41 ha Waite Amulet dam contains approximately 5.9 Mt of tailings with a 60-90% sulphide composition, pyrite being the main mineral. Tailings were deposited from 1933 to 1962 with the exception of one shutdown period.

In the late 1960's, surface water quality monitoring revealed evidence of acid generation due to the oxidation of iron sulphides. Neutralization of acidic water was initiated at that time. By 1969, the Horne Division had decided upon the need for a vegetative cover to control tailings dusting and acid generation as well as to improve site aesthetics.

Surface pH of the tailings ranged between 2.2-3.0. Starting in 1969, an experienced contractor undertook the reclamation work. Lime at 11.2 t/ha, fertilizer and seed were added based upon experience at other sites and although a green tinge developed, the vegetation died back within a month. A second attempt was made the next year with a higher limestone addition rate (15.7 t/ha) with an unfortunate similar result - total failure. Noranda then decided it was time for a new approach.

Limestone neutralization tests were conducted in the laboratory and a tailings acid demand of 56 t/ha of lime was indicated. In 1972, 10 100 m^2 plots were established and 112-168 t/ha of limestone and varying fertilizer and seed mixtures were applied. Although results were promising, costs were quite high. Variability in test plot performance introduced some technical uncertainty and consequently the Horne Division was reticent about proceeding with another full scale reclamation attempt.

3.0 Noranda/University of Guelph Program - Vegetation

Noranda Inc. initiated a five year research program with the University of Guelph, beginning in 1974, to evaluate the variability of sulphide tailings and how this could affect plant growth and to improve the technique for assessing growth potential of these sulphide materials. Although copious data were generated, correlations between growth potential and chemical and physical characteristics of tailings were not evident! Conventional metallurgical and agricultural analyses did not elucidate the problem. What did develop from the research program, however, was a rapid growth room assessment technique whereby a relatively large number of tailings samples with varying applications of lime, fertilizer and seed could be evaluated. This technique has subsequently been applied within Noranda Inc. companies and some reclamation projects based on these laboratory scale assessments have been successfully carried out.

Waite Amulet was vegetated in stages from 1978 to 1981 using agricultural limestone at a rate of 45 t/ha and lush, self-sustaining and maintenance free vegetation exists to date.

4.0 The Effectiveness of Waite Amulet Vegetation

Stream water quality monitoring on and adjacent to the Waite Amulet site continued and in 1982 an expanded sampling program was undertaken. The news was not good. Acidity, iron, zinc and copper concentrations averaged as high as 8000, 2400, 115 and 14 mg/L respectively at some individual sampling stations located on four streams draining the site and pH generally ranged between 2.5-3.0. An improvement in water quality has not been evident.

Qualitatively, the vegetative cover has been effective in controlling fugitive dust and surface water erosion, as well as in improving site aesthetics. Indeed, these objectives have been fully achieved. A reduction in the rate of acid generation which can be ascribed to vegetation is believed to be minimal, if any took place at all. An active, water consuming surface cover will tend to depress the water table and consequently the potential for oxygen penetration is enhanced. However, it must be stressed that baseline data collection was inadequate for a proper, quantitative evaluation of acid generation and water balance after reclamation was completed.

By 1984, an acid storage pond and a modern waste water treatment plant were constructed at a cost of 1.8 M$. The lime neutralization facility has been operating efficiently on a seasonal basis at a cost of 0.2 M$ annually.

5.0 The Current Noranda Inc. Research Effort

With such a significant inventory of reactive tailings requiring decommissioning, practical and economical technology to control acid generation needs to be developed. The number of properties approaching the decommissioning phase within the next five years adds a sense of urgency to the quest for a better understanding of processes involved in acid generation and for environmentally efficient and reasonably cost effective technology.

Consequently, Noranda Inc. initiated discussions with the federal Department of Energy, Mines and Resources (CANMET) and other mining companies to formulate a national research program. The Reactive Tailings (RATS) program, now funded equally between the sponsoring industry and CANMET, began to evolve in 1984.

Currently, Noranda Inc. is sponsoring research on two fronts - first, a RATS study at the Waite Amulet tailings site being conducted by Centre de Recherche Noranda and CANMET and second, a University of Waterloo, Institute for Groundwater Research program. In general, the broad objectives of these efforts are to:

- characterize the geochemistry within reactive tailings;

- identify the processes which control the geochemistry;

- determine, evaluate and demonstrate the mechanisms which control the geochemical reactions; and

- predict the geochemical behaviour of reactive tailings under various reclamation scenarios.

Several technical papers focussed on these broad objectives have already been published on the Waite Amulet study program by Siwik and Dave, 1987, and Dave et al. 1986, and others will follow. A series of papers will also be generated by D.W.Blowes on the University of Waterloo program.

The technical discussion which follows includes information from the Waterloo investigations as they pertain to the vegetation/acid generation relationship.

The sites included here are:

- Heath Steele Mines Limited, Newcastle, New Brunswick (managed by Brunswick Mining & Smelting Corporation Limited)

- Waite Amulet Tailings Site, Rouyn-Noranda, Quebec (managed by the Horne Division)

- a high carbonate tailings site in Ontario, formerly owned by a Noranda Inc. company; and

- Nordic Mine Tailings, Elliot Lake, Ontario, not a Noranda Inc. site but one researched by the University of Waterloo.

The field methods utilized by the University of Waterloo are outlined in Appendix A. A description of geochemical processes is provided in Appendix B.

5.1 Heath Steele Mines Limited, Newcastle, New Brunswick

The 4 ha tailings pond at Heath Steele Mines Limited was examined by Boorman and Watson, 1976 of the New Brunswick Research and Productivity Council. Although no reclamation work has been initiated on these tailings, the location was included in the University of Waterloo program to examine the geochemistry ten years after the earlier research effort. Deposition of tailings containing 85% sulphide minerals, mostly pyrite and pyrrhotite, was discontinued in 1964. Pore water chemistry in the vadose and groundwater zones was determined as well as gas concentrations in the unsaturated tailings layer.

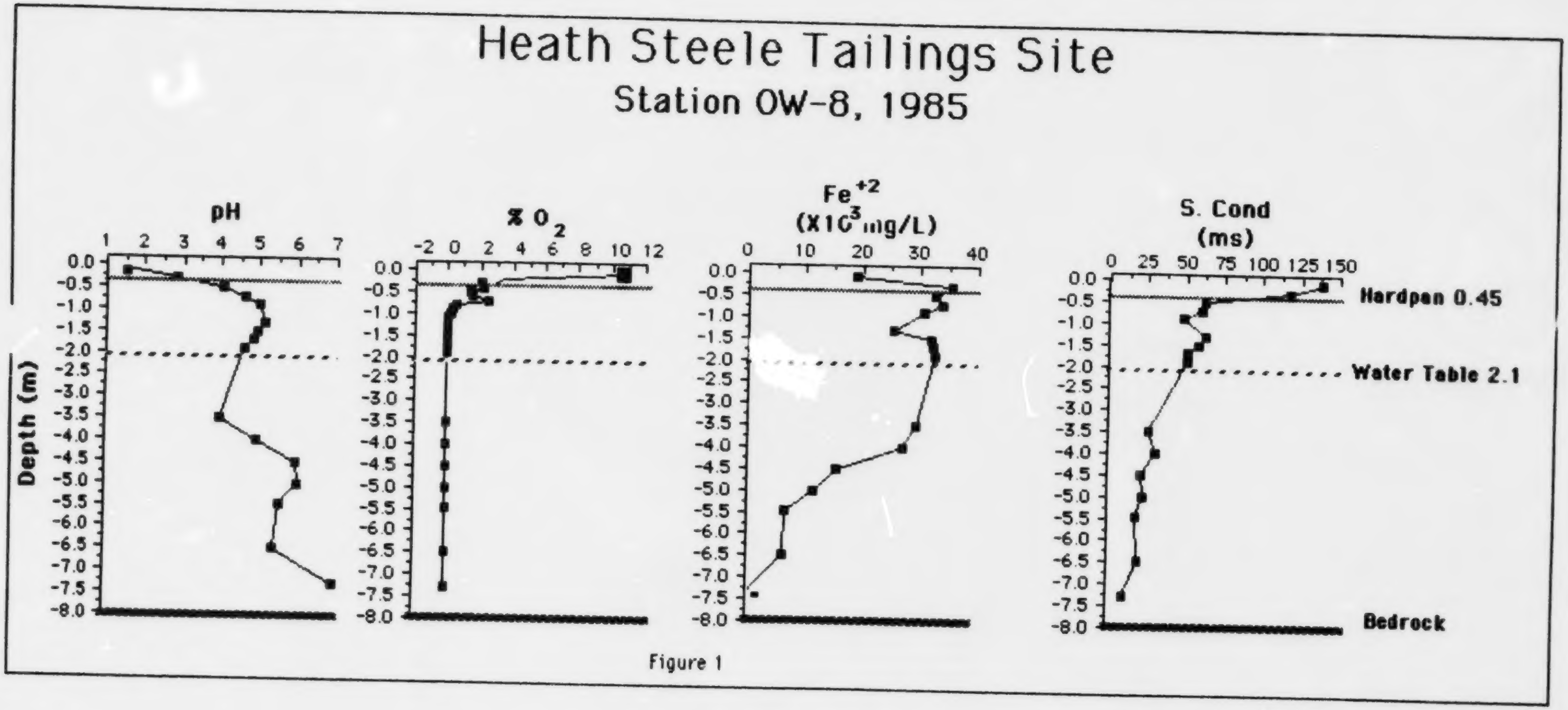

Heath Steele Tailings Site
Station OW-8, 1985
pH
% O2
Fe+2 (X10³ mg/L)
S. Cond (ms)
Depth (m)
Hardpan 0.45
Water Table 2.1
Bedrock
0.0
-0.5
-1.0
-1.5
-2.0
-2.5
-3.0
-3.5
-4.0
-4.5
-5.0
-5.5
-6.0
-6.5
-7.0
-7.5
-8.0
1 2 3 4 5 6 7
-2 0 2 4 6 8 10 12
0 10 20 30 40
0 25 50 75 100 125 150

Figure 1

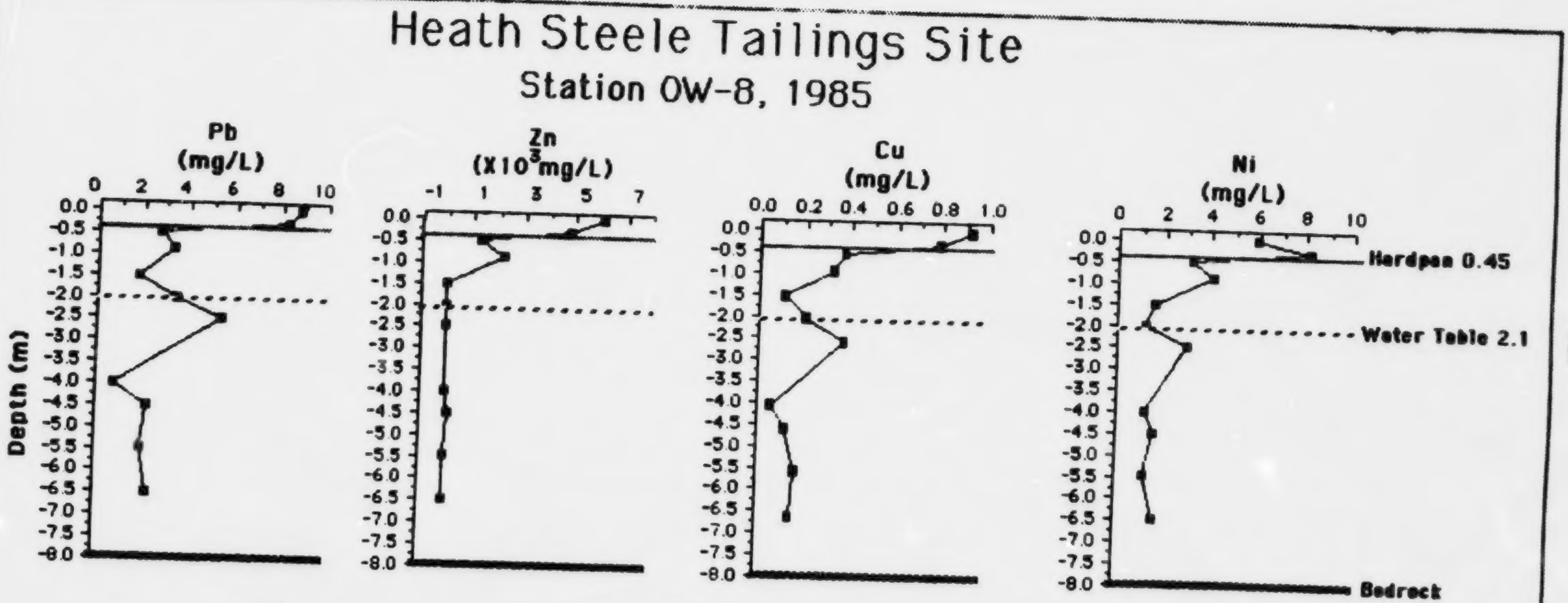

Heath Steele Tailings Site
Station OW-8, 1985
Pb
(mg/L)
0 2 4 6 8 10
Zn
(X10^{3}mg/L)
-1 1 3 5 7
Cu
(mg/L)
0.0 0.2 0.4 0.6 0.8 1.0
Ni
(mg/L)
0 2 4 6 8 10
Depth (m)
0.0 -0.5 -1.0 -1.5 -2.0 -2.5 -3.0 -3.5 -4.0 -4.5 -5.0 -5.5 -6.0 -6.5 -7.0 -7.5 -8.0
Hardpan 0.45
Water Table 2.1
Bedrock

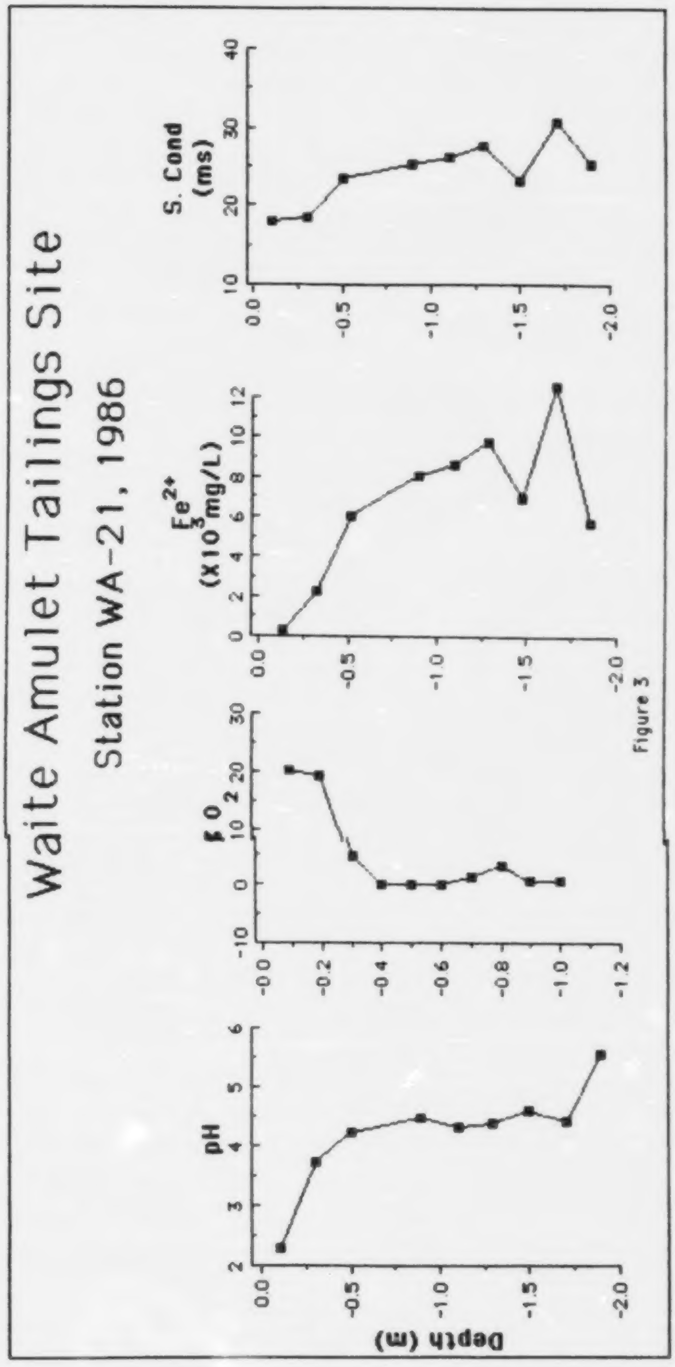
Figure 2

Low pH and extremely high levels of Fe^{2+} and specific conductance were recorded at this site (Figure 1). The low pH, high Fe^{2+} conditions were observed to be directly related to oxygen penetration depths. Very high concentrations of several metals (Cu, Pb, Zn, Ni) were observed near the surface in the vadose zone (Figure 2). These metals were probably released to solution by the oxidation of sulphide minerals. This tailings impoundment has not had the benefit of a vegetation program, but the processes controlling pore water evolution appear to be the same as at the other sites investigated. Very high concentrations of metals in solution are attributed to the high sulphide content of the tailings.

5.2 Waite Amulet Tailings, Rouyn-Noranda, Quebec

The Waite Amulet tailings impoundment is an ideal location for investigating the geochemistry within a reactive tailings mass since the site has barren, directly vegetated and clay capped and vegetated regions.

Results from the Quebec field site indicate that pyrite oxidation has generated low pH, high Fe^{2+} and high specific conductance levels in the vadose zone of the tailings. Near the surface and within the unsaturated zone, the pore water pH level is generally below 2.5. Redox potential is quite



Waite Amulet Tailings Site

Station WA-21, 1986

Figure 3

high, demonstrating an abundance of oxidized species in solution at shallow depths. Concentrations of Fe^{2+} increase in the vadose zone as illustrated in Figure 3 as O_2 is consumed in the pyrite oxidation reactions. The solution pH increases with depth as the O_2 concentration declines. Similar changes in tailings pore water geochemistry were observed at each sampling location. The depth of the low pH, high redox potential zone was observed to increase with O_2 penetration which in turn was a function of the distance to the water table. Pore waters with high Fe^{2+} concentrations and specific conductance have been displaced downwards past the water table by the gradual infiltration of rainwater and snowmelt.

5.3 A High Carbonate Tailings Site, Ontario

Tailings containing approximately 4% sulphide and 20% carbonate minerals were deposited from the early 1940's to the early 1960's. The mill at this site extracted gold from the sulphide-bearing, carbonate-rich ore. A successful vegetation program was initiated in 1971 on the 30 ha tailings surface.

Investigations included pore water sampling in the unsaturated and saturated zones and gas sampling in the vadose zone at several locations on and adjacent to the tailings pond.

High Carbonate Tailings Site

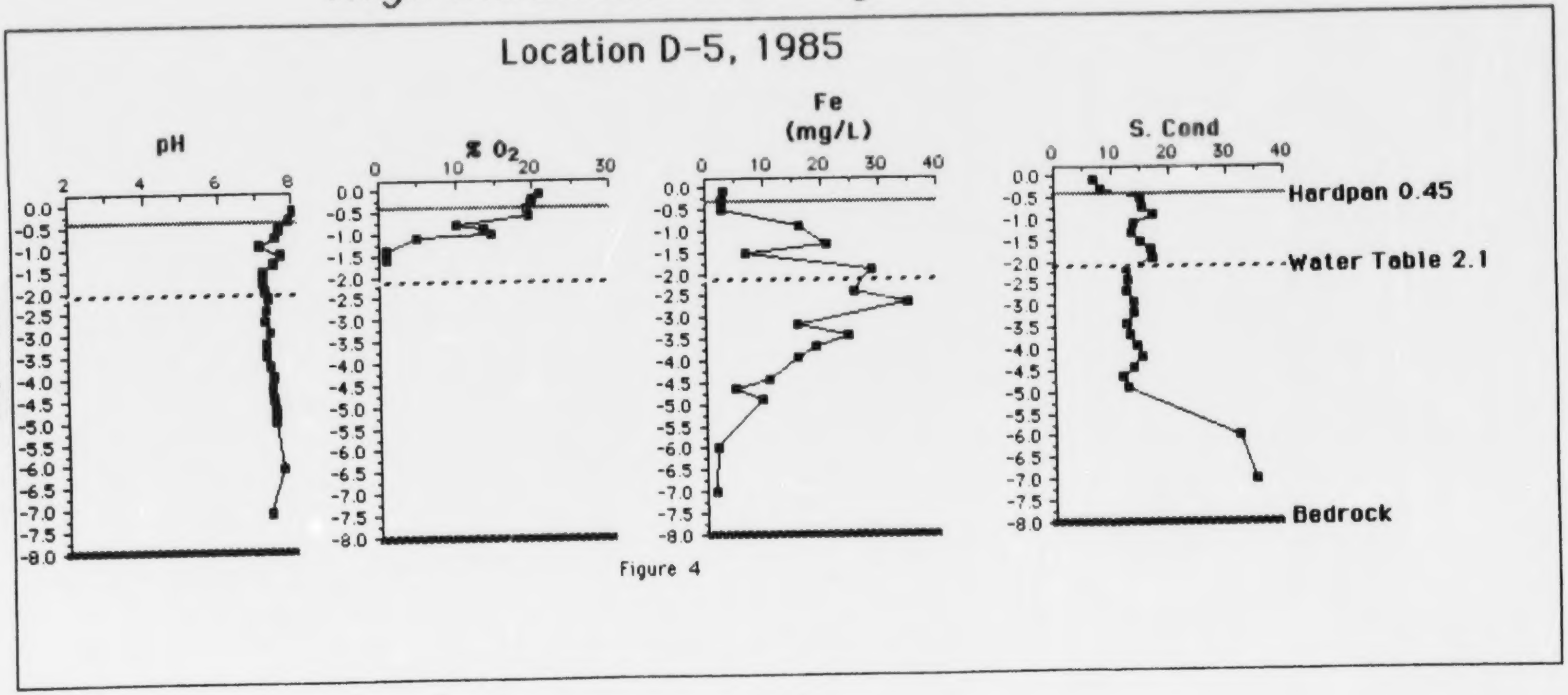


Figure 4

High Carbonate Tailings Site

Location D-5, 1985

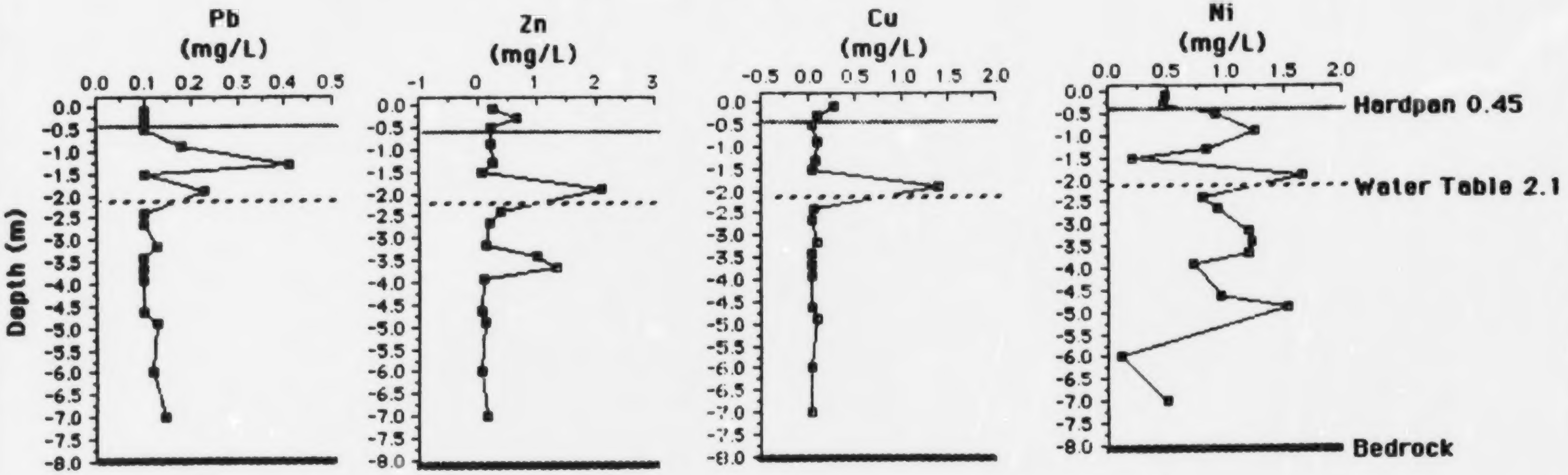


figure 5

No acidic pore waters were detected. High concentrations of Fe^{2+} and specific conductance observed in the vadose zone indicate pyrite oxidation is occurring (Figure 4). The high pH levels suggest that the buffering capacity of the carbonate minerals and the rate of carbonate dissolution are sufficient to consume the H^+ generated by pyrite oxidation. Concentrations of other metals (Cu, Pb, Zn, Ni) exist at shallow depths in the unsaturated zone (Figure 5). It is likely that these elements were released to solution as a result of sulphide mineral oxidation.

5.4 Nordic Mine Tailings, Elliot Lake, Ontario

The Nordic Mine tailings, composed mainly of silicate minerals - approximately 5% pyrite and 0.06% calcite - were deposited from 1957 to 1968. A successful vegetation cover was established between 1973 and 1978. Blair et al., 1980, Dubrovsky et al., 1981, Smyth, 1982, Morin et al., 1982 have investigated the vadose and groundwater zones at several locations on and adjacent to these tailings.

The results of the investigations are summarized by Dubrovsky et al. (1985). These authors observed that low pH, high Fe^{2+} and $SO_4{}^{2-}$ waters are present at shallow depths at each of the sampling locations. Dubrovsky (1986) suggests that the high Fe^{2+}, $SO_4{}^{2-}$ pore waters are being displaced downwards through and out of these tailings by infiltrating rainfall and snowmelt at 0.24 to 2.0 m/a (average of 1.0 m/a), and that the low pH pore waters are being neutralized by carbonate consuming reactions.

Smyth (1982) concluded that the effects of pyrite oxidation were greatest at locations where the water table was deep and the tailings were coarse and, therefore, where O_2 diffusion would be greatest.

6.0 Discussion

Oxidation of sulphide minerals was evident in the upper 1 to 2 m of the vadose zone in the four tailings impoundments which were investigated. The extent of oxygen diffusion into the tailings is dependent on the depth of the water table and the sulphide content and particle sizing of the tailings. It appears to be independent of the presence of vegetation. Tailings with high pyrite content consume O_2 rapidly, and, therefore, the O_2 penetration depth is limited by this factor.

The oxidation of sulphide minerals generates low pH conditions and releases various constituents, principally Fe^{2+} and SO_4^{2-}, as well as concentrations of Cu, Pb, Zn, Ni and As. High concentrations of Fe^{2+} represent a large potential source of acidity; each mole of Fe^{2+} will generate two moles of H^+ as a result of oxidation and precipitation of ferric hydroxides.

The H^+ generated by pyrite oxidation is neutralized by H^+ consuming reactions - the dominant process being the dissolution of carbonate minerals as described by the reaction

$$CaCO_3 + 2H^+ <==> H_2CO_3 + Ca^{2+}$$

under low pH conditions. This process can raise the pH to neutral or basic levels at which many metal concentrations will be controlled by hydroxide solubility constraints. Metal levels not influenced by precipitation-dissolution reactions may be controlled by adsorption-desorption processes on the surfaces of tailings solids and by co-precipitation. The mechanisms will depend on the bulk solution chemistry and on pH conditions. The overall effect of carbonate consuming reactions is to retard the movement of low pH fluid and, in many cases, high metal concentrations relative to the movement of infiltrating water through and out of the tailings impoundment.

At the high carbonate tailings location, the neutralization capacity was large enough to maintain neutral to alkaline pH conditions throughout the mass, including the vadose zone.

The elevated pH at this site limited metal concentrations to much lower levels than were observed at any of the other higher sulphide tailings ponds.

Pore water is being displaced downwards through the vadose and into the groundwater zone at each location due to infiltrating rainfall and snowmelt. Pore waters containing Fe^{2+} and SO_4^{2-} will eventually be displaced out of the tailings impoundment, either into the underlying groundwater flowsystem or into the surface seepage system. In the Nordic Mine tailings, the high Fe^{2+} and SO_4^{2-} pore waters have entered the underlying aquifer (Dubrovsky et al., 1985) while at Waite Amulet the discharge is via seepage along the flanks of the pond.

7.0 Conclusions

A self-sustaining vegetative cover can be established directly on reactive tailings. Rapid techniques for assessing the application rate of fertilizer, seed and alkali and for designing a vegetation program have been developed. Vegetation effectiveness in controlling acid generation, however, is limited, if not negligible. Based on preliminary information evolving from the RATS and the University of Waterloo research, the following conclusions have been formulated:

7.1 Pyrite oxidation is occurring within tailings containing 4-90% sulphide minerals independent of a vegetative cover. Metals such as Fe^{2+}, Zn, Pb, Cu, Ni, and As, and SO_4^{2-} are being released to the pore water in this oxidative zone.

7.2 Processes of neutralization, precipitation, co-precipitation and adsorption also occur within tailings below the reactive zone. The data from the four tailings sites illustrate a decrease in some metal concentrations and an increase in pH as the pore water is displaced downwards. With time, this flow will report either to groundwater or to surface seepage.

7.3 The zone of active sulphide oxidation is a direct function of the depth of oxygen penetration. This may be controlled by the depth of the water table and by the sulphide content and particle sizing of the tailings. Significant quantities of reactive tailings remain above the water table at each of the sites and as these sulphides are consumed, the downward flow of reaction products will continue.

7.4 Direct vegetation of reactive tailings does not restrict pyrite oxidation nor does it limit water infiltration sufficiently to significantly reduce water flow from the impoundment. Direct vegetation may cause the average water table to recede during the growing season and, therefore, the total mass of sulphides available for oxidation may increase.

7.5 The rate of acidity and heavy metal discharge from a reactive tailings mass cannot be predicted or modeled based upon the existing level of understanding of the geochemistry. Ongoing research efforts within RATS and the University of Waterloo program will address these unknowns.

Appendix A

Field Methods

Pore water from the tailings vadose zone was sampled using the modification of the Patterson et al. (1978) technique described by Smyth (1982). Samples were obtained at approximately 20 cm intervals throughout the unsaturated layer.

Vadose zone pore gas samples were collected using the method described by Reardon and Poescente (1984), and analyses were determined using a portable meter (Nova model 305LBD). Pore gas samples were also returned to the laboratory and analyzed by gas chromatography.

Water samples from the groundwater zone were collected using modified Cassagrande drive point piezometers of the type described by Dubrovsky et al. (1981). Samples were collected using the syringe sampling technique of Gillham (1982).

Pore water samples retained for analyses by atomic absorption were acidified immediately with concentrated HCl and refrigerated. Ferrous iron was determined in the field using the potassium dichromate method (Waser, 1966). The pH and redox potential of the pore waters were measured using a liquid filled combination pH electrode (Orion Ross) and a combination platinum electrode (Orion 96-78). Electrodes were calibrated using pH 1.7, 4.0 and 7.0 buffers which were maintained at groundwater temperatures. Measurements of pH and redox potential in the saturated zone were made in closed flow-through cells also maintained at groundwater temperatures.

Appendix B

Geochemical Processes

Geochemical reactions which control pore water chemistry are oxidation-reduction, dissolution-precipitation, coprecipitation and adsorption. The dominant reaction in the geochemical evolution of tailings pore water is the oxidation of pyrite to produce low pH conditions and high concentrations of Fe^{2+} and SO_4^{2-}. The oxidation of pyrite may be described by the reaction

$$FeS_2 + 7/2\ O_2 + H_2O \dashrightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+$$

Nicholson (1984) suggested that in tailings environments the rate of pyrite oxidation is limited by the rate of oxygen diffusion into the tailings impoundment.

Ferrous iron in solution may be further oxidized to ferric iron through the reaction

$$Fe^{2+} + 1/2\ O_2 + H^+ \dashrightarrow Fe^{3+} + H_2O$$

Ferric iron can then precipitate as ferric hydroxide to release an additional three moles of hydrogen ion through the reaction

$$Fe^{3+} + 3H_2O \dashrightarrow Fe(OH)_3 + 3H^+$$

Ferric iron can also oxidize additional pyrite or other sulphide minerals through reactions of the form

$$2MeS + 7/2\ O_2 + H_2O \dashrightarrow 2Me^+ + 2SO_4{}^{2-} + 2H^+$$

where Me represents a metallic ion. Oxidation of pyrite and other sulphide minerals releases additional metals, particularly Cu, Ni, Pb, Zn and As into solution. These oxidation products are transported by the physical processes of advection, dispersion and molecular diffusion. Concentration of reaction products is also controlled by the geochemical processes of precipitation and dissolution, adsorption and desorption, and co-precipitation.

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TREATMENT OF ACID MINE WATER AND THE DISPOSAL OF LIME NEUTRALIZATION SLUDGE

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ABSTRACT

The standard procedure for treating acid mine drainage is neutralization with lime and precipitation of the metals as hydroxides. The treatment process including chemical precipitation, oxidation, and solids/liquid separation is described. Alternative technologies are briefly addressed. The nature of the sludge, generated from acid mine water treatment, is affected by a number of physical and chemical parameters which depend on both the characteristics of the effluent and the treatment process. Methods of settling, densifying and dewatering sludges are described, and long-term storage and stability are discussed. Further work regarding precipitation chemistry, in-situ dewatering of low density sludges and decreased leachability of sludges is recommended.

INTRODUCTION

Acid mine water (AMW) occurs at approximately one-third of the mines in Canada as well as at other locations. This acidic wastewater results from the oxidation of sulphide material found in sulphidic mineral formations within and around mines, tailings, and waste rock dumps, and with reactive slate rock. Treatment is necessary because the wastewater is highly acidic and contains high concentrations of metals, sulphate, and in some cases, radionuclides.

This paper presents the current "State-of-the-Art" for treatment of AMW to control the acidity, high concentrations of iron and heavy metals and disposal of the resultant sludge. Information available in the literature will be presented, along with data from full scale operating treatment plants, in particular, Noranda Group facilities.

TREATMENT OF ACID MINE WATER

Overview

The treatment of acid mine water involves, firstly, the addition of an alkali to increase the pH to neutralize the acidity and precipitate metals, secondly, facilities for solid/liquid separation to remove metal hydroxide and gypsum precipitates, and finally, the disposal of the resulting metal-containing sludge. Oxidation of ferrous iron is generally incorporated in the process to facilitate iron removal and provide a lower solubility product. The optimum pH for neutralization and effective metal removal is dependent on the metal species present and may range from 6.5 to 11.5.

Alternative technologies to lime neutralization have been considered for the treatment of AMW. These methods, which include reverse osmosis, ion exchange, and water softening, are briefly reviewed in this paper. The techniques were assessed in laboratory studies and were generally designed for the treatment of coal mine drainage. They have not been applied to AMW treatment at full scale since they may not be cost effective.

The use of reverse osmosis (RO) in the treatment of AMW has been studied by the US EPA (Escher et al., 1983). The purpose was to produce potable water which will be low in total dissolved solids (TDS). During treatment, the potential for iron fouling and gypsum formation was minimized by keeping the pH between 2.8 and 3.0. Tests were performed on a 15 m^3/d, once-through RO unit, at an average flux rate of 500 $L/m^2/d$ operating at 29.1 kg/cm^2. A 75% recovery of clean water was realized. The resulting highly contaminated concentrate must be treated, usually by lime neutralization;

although the volume of water to treat would be significantly reduced.

Like RO, ion exchange (IX) will produce a high quality effluent and a concentrate stream (regenerant solution) needing treatment. Several processes were reviewed by Escher et al.,(1983). One is the 'Sul-bisul Process' which employs a 2 or 3 bed system to remove ferrous sulfate. Cations in the treated water are exchanged by a strong-acid resin, the effluent is decarbonated, and then passed through a strong-base anion resin to remove sulphate. This process is effective for AMW with TDS less than 3000 mg/L. Costs are likely \$0.20 to 0.25/m^3 (US\$, 1980).

Another IX method, the 'Modified Desal Process' uses a weak-base anion resin to remove sulphate and convert metals to bicarbonate salts. The metal bicarbonate solution is then aerated and lime neutralized to remove metals. This system adds an extra step to AMW treatment but effects sulphate removal and eliminates subsequent gypsum formation. Operating costs (including capital depreciation) were \$0.48/m^3 (US\$, 1975). The third method is the 'Two Resin Process' which uses a strong-acid cation exchanger to remove metals, followed by a weak-base anion exchanger to remove the resulting sulphuric acid. Operating costs appear to be about 50% more than the Modified Desal system.

Chemical softening is employed to remove dissolved ions from AMW if water is to be considered for industrial use or as potable water. The alumina-lime-soda process is applied to raw AMW and is applied to waters mainly containing sulfate. It is capable of removing metals and hardness. Sodium aluminate and lime neutralize acidity, precipitate heavy metals and remove hardness and gypsum. Capital costs for a 3800 m^3/d plant were \$516 000 (US\$, 1975) with operating costs of \$0.24/m^3 (Escher et al., 1983).

Neutralization/Precipitation

Chemistry - The treatment of AMW is usually with an alkali (eg. lime) that results in the neutralization of the acidity and the precipitation of the dissolved metals in the hydroxide form. Oxidation of ferrous to ferric iron is also common. These three processes are interrelated, but for the sake of simplicity, will be discussed separately in the following paragraphs.

AMW neutralizing reagents that could be considered include lime (CaO, $Ca(OH)_2$), limestone ($CaCO_3$), and caustic soda ($NaOH$). Lime is the most commonly used alkali because of its high reactivity, its availability and its relatively low cost. Lime is applied in all current operations in Canada. The reaction with lime and AMW is presented in equation (1).

$$4Ca(OH)_2 + Fe_2(SO_4)_3 + H_2SO_4 \rightarrow 4CaSO_4 + 2Fe(OH)_3 + 2H_2O \quad (1)$$

All heavy metals will generally precipitate similarly to ferric iron. Gypsum ($CaSO_4$) forms during lime neutralization and can result in significant problems during the treatment process. This scaling potential will be briefly discussed later.

The use of limestone in lieu of lime in the treatment of AMW was assessed in studies described by Wilmoth (1977) and McDonald et al.,(1981). Although limestone is about 35% of the cost of lime, its use is restricted due to limitations that include a slow reactivity, an inability to raise the pH much above 6.0, poor dissolution, poor effluent quality after treatment, and high transportation costs. Lime and limestone in a two stage process was found to effectively treat AMW (McDonald et al., 1981). Limestone was used at lower pH ranges (3.4 to 4.1) and lime was used in the pH 6.0 to 7.0 range to complete the treatment process. On the other hand, Huck et al. (1978) found that two-stage neutralization using limestone/lime treatment did not result in lower metal concentrations and showed no cost advantage over the use of lime alone.

The merit in using NaOH is to eliminate gypsum formation, but much higher dissolved solids concentrations will be observed due to the high solubility of sodium sulphate. However, the high cost and undesirable handling properties of NaOH have limited its use.

Sulphide precipitation has also been investigated for AMW treatment, although, this is less common. A two stage process of chemical addition to mine drainage was successful in removing dissolved metals (Larsen et al., 1973). The first stage consisted of lime addition to pH 5.0 with iron and aluminum being largely precipitated while other metal species were left in solution. The second stage involved sulphide addition to precipitate the other metals; with final pH adjustment to pH 6.5. Drawbacks in the use of sulphide include the high cost of reagents, difficulties in chemical dosing, and a high potential for toxic sulphide ions in the effluent.

To assess alkali dosage requirements lab neutralization tests should be performed. These tests must address the seasonal variability of AMW quality and the resultant impact on lime requirements. The theoretical alkali requirement for AMW neutralization can be determined as follows:

$$\text{Alkali needs} = \text{total acidity}/(E \times F) \qquad (2)$$

where total acidity = standard lab analysis for acidity expressed as $CaCO_3$; E = expected efficiency of alkali (eg. hydrated lime @ 0.7); and F = $CaCO_3$ equivalence of alkali (hydrated lime = 1.35; quicklime = 1.78).

Heavy metal hydroxides form amphoteric precipitates (ie. have a minimum solubility at a distinct pH). Figure 1 presents the hydroxide solubility curves for the metals commonly found in AMW. Optimum precipitation for ferrous and ferric iron occur at a very different pH. Ferric iron and aluminum will be effectively removed at a relatively low pH ranging from about 6.5 to 8.0, while for removal of ferrous iron and other common metals higher pH values are required. The addition of lime (or other alkali) to raise the pH to about 9.5 is the most common method for the neutralization of AMW. At this pH, effective precipitation of all metal species normally found in AMW occurs. The optimum pH, for three wastewaters tested at pilot scale, was found to be 9.5, although effluent quality was not adversely affected by a wider range of 8.9 to 10.5 (Huck, et al., 1974; 1978).

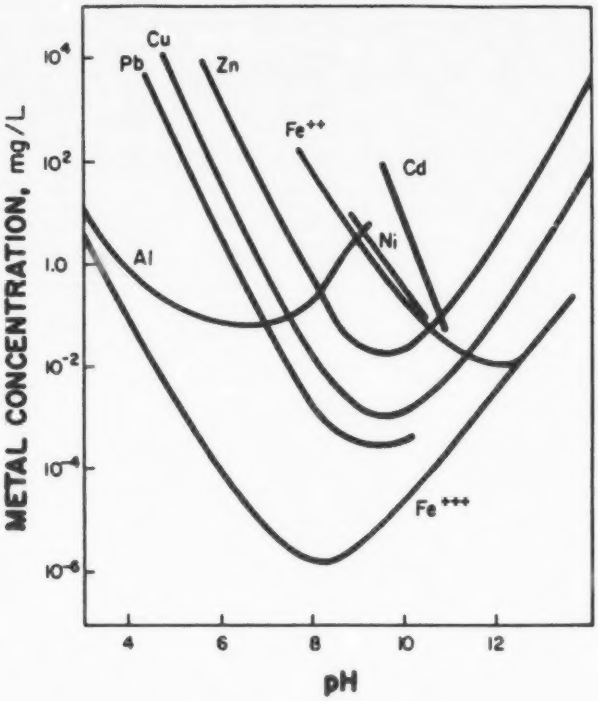


Figure 1. Metal Hydroxide Solubilities

In cases where iron and aluminum are the predominant species, but other heavy metals are present as well (e.g. reactive slate), two stage neutralization may be considered. In the first stage, pH adjustment to about 6.0 will effect aluminum and ferric iron precipitation. (An oxidation step could be included to convert ferrous iron to ferric, if needed). Following solids removal, second stage precipitation at approximately pH 9.0 should ensure complete removal of other metal species.

Oxidation - It is advantageous to oxidize ferrous iron to ferric iron and thus remove ferric iron which will form a more stable precipitate. As illustrated in Figure 1, ferric hydroxide will remain insoluble over a very broad pH range, while ferrous hydroxide will rapidly solublize at pH values less than 11.0. The oxidation rate depends on ferrous iron concentration, pH, dissolved O_2, and temperature. The oxidation of ferrous iron occurs in minutes at a pH greater than 7.5, but requires hours at pH 6.5 or less. Figure 2 illustrates that ferrous iron can be fully oxidized in minutes depending upon the initial concentration and neutralizing pH.

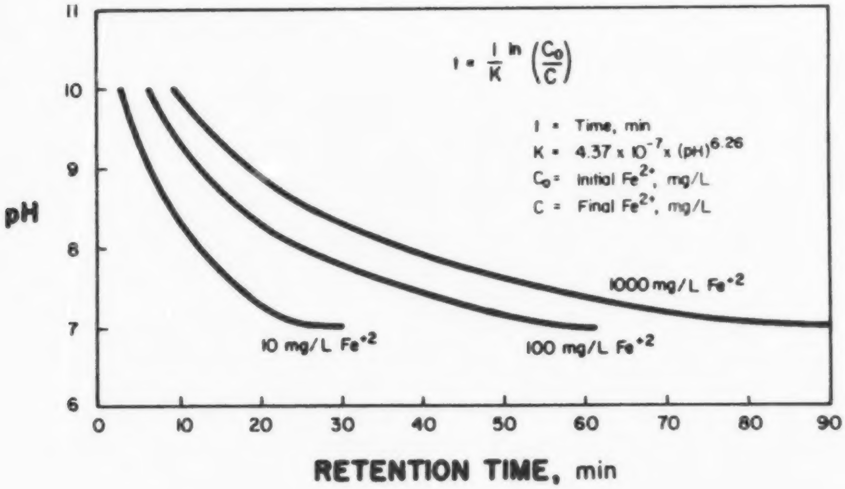


Figure 2. Retention Time vs pH for Ferrous Iron Oxidation

The stoichiometric requirement for oxygen in the conversion of ferrous iron to ferric iron is 0.14 kg O_2 per kg Fe. Because oxygen makes up about 21% of air by volume and only a fraction of the oxygen that comes into contact with water is actually absorbed, the theoretical demand for air can be expressed as:

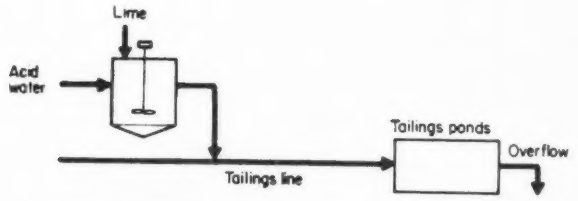
$$A = 0.196 x Q x C/E \qquad (3)$$

where A = total air demand (m^3/h)
Q = AMW flow (m^3/d)
C = ferrous iron concentration (mg/L)
E = oxygen transfer efficiency (%)

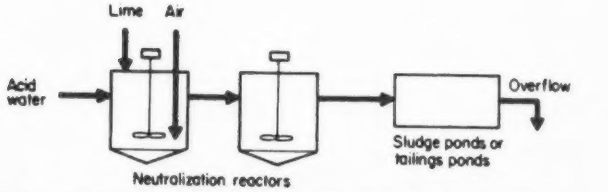
While oxygen transfer efficiencies range from 3 to 15%, a design value of 5% is normally used when manufacturers' data are not available.

Other oxidants such as ozone, hydrogen peroxide, chlorine, and potassium permanganate are generally too expensive to be applied to AMW oxidation although some applications may be feasible. Ozone could be used at pH 6.0 where reaction times may be short, allowing the use of smaller reactors. Hydrogen peroxide merits consideration only under specialized conditions; while chlorine and permanganate are impractical to use. Biological oxidation, using a rotating biological contactor (RBC) or trickling filter, can also be applied to oxidize ferrous iron to ferric iron. Although, for a RBC system, the capital cost is greater than for mechanical aerators, the operating costs are significantly lower because of reduced energy requirements.

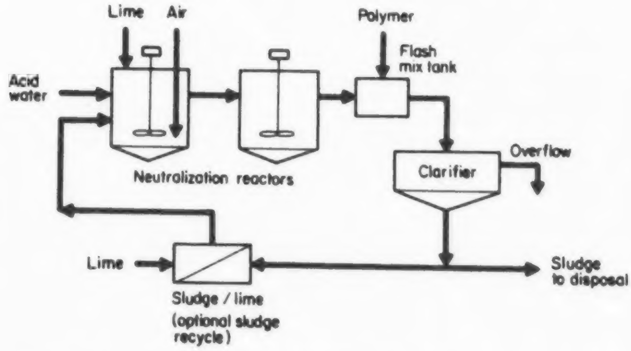
Process - Neutralization of AMW is generally accomplished with lime. Hydrated lime is applied at most operations in Canada, but for liming requirements of over 2000 t/a, quick lime is recommended because of economics. Treatment can be conducted in one of three ways (Figure 3). The simplest method is to neutralize AMW with lime which is then pumped with the tailings slurry to a tailings pond for solid/liquid separation. In a second method, lime addition to the AMW in a reactor, with aeration to oxidize ferrous iron, provides better control of the neutralization stage. Solid/liquid separation is provided by settling or tailings ponds. The most complex method is the lime neutralization plant that includes reactors and mechanical clarifiers which provide control of both the neutralization and solids/liquid separation stages. The treated effluent from the above processes will eventually be discharged to a receiving water and the removed solids (sludge) disposed of by various means (which will be discussed later).


Lime
Acid water
Tailings ponds
Overflow
Tailings line

Type 1. Lime addition to effluent or tailings lines.


Lime
Air
Acid water
Overflow
Sludge ponds or tailings ponds
Neutralization reactors

Type 2. Lime addition to neutralization reactors. Solid/liquid separation in ponds.


Lime
Air
Polymer
Flash mix tank
Acid water
Overflow
Clarifier
Neutralization reactors
Lime
Sludge / lime
(optional sludge recycle)
Sludge to disposal

Type 3. Lime addition to neutralization reactors. Solid/liquid separation in clarifiers.

Figure 3. Types of Noranda Group Treatment Facilities

Effective treatment of AMW depends upon providing adequate retention time in the reactors to neutralize the wastewater and ensure complete oxidation of ferrous iron and hydrolysis of the metals. Retention time is dictated by the time necessary for lime dissolution, air dispersion, the increase in pH, the oxidation of ferrous iron to ferric iron, and metal hydrolysis. The general practice is to design for a minimum of 20 minutes retention time in the reactors to ensure complete lime neutralization and aeration.

Providing sufficient mixing energy (power) in the neutralization reactor is essential to disperse the neutralizing reagent, ensure complete reaction and maintain the precipitates in suspension. The main power requirements will be during aeration to ensure good dispersion of the air bubbles and effective diffusion of oxygen into the solution.

Air is commonly added to the neutralization reactor by air sparging. An alternative to neutralization reactors would be an in-line aeration and neutralization system (ILS) (Ackman et al., 1985; 1986). The unit consists of a jet pump (venturi configuration) to entrain air and a static mixer to induce turbulent flow. Several jets can be used in parallel as long as a minimum water pressure of 1.41 kg/cm^2 per jet is maintained. Field tests (Ackman et al., 1985) and recent Noranda lab studies have shown that single stage oxidation was effective in oxidizing AMW with less than 200 mg/L ferrous iron.

In many cases during lime neutralization, scaling occurs as a gypsum deposit due to elevated sulphate concentrations. Based on Noranda's experience, scaling tendencies can be correlated to influent sulphate concentrations, which are summarized as:

Influent Sulphate	Scaling Tendency
<3000 mg/L	none to mild
3000 to 5000	mild to moderate
5000 to 6000	moderate to severe
>6000	severe

Huck et al. (1974) found that at sulphate concentrations less than 2500 mg/L, gypsum deposition was found to be minimal; at concentrations greater than 3000 mg/L, scaling occurred rapidly. As well, at concentrations greater than 12 000 mg/L, pellets as large as 0.64 cm in diameter were found to form.

Solid/Liquid Separation

Clarification - Solids/liquid separation is usually accomplished by settling the solids either in a tailings impoundment, settling pond or in a clarifier/thickener (clarifier). Economics and land availability generally dictate the type of settling device to be used. The majority of mining operations use the tailings impoundment for sludge settling and disposal.

Settling ponds are designed primarily for solids/liquid separation, with periodic sludge removal. The normal concentrations of solids in AMW sludge is 1 to 2%, although values greater than 2.5% have been reported (Ackman, 1982 and Escher et al., 1983). Settling ponds fail when accumulated sludge quantities reduce pond volumes sufficiently so that effective solids/liquid separation does not occur. Because of the thixotropic nature of AMW sludge, significant resuspension of solids can occur by wind action on the ponds resulting in poor effluent quality.

A clarifier is a gravitational solids/liquid separator having the primary function of producing a high quality supernatant. Mechanical clarifiers are preferred technology when land availability is limited, particularly when no tailings pond is present. The upflow clarifier serves as a multifunctional unit with flocculation, sedimentation, and filtration occurring within the same vessel. An inclined-plate, shallow-depth settler (lamella type) performs the same function as a conventional clarifier but occupies approximately 10% of the space. However, a major disadvantage of this unit is its suseptibility to clogging with gypsum and the significant reduction of clarifier volume for sludge thickening.

Loading rates for clarifiers and ponds that do not use sludge recycle as part of the treatment process were reported to be in the range of 0.18 to 0.35 $m^3/m^2/d$ (Escher _et al_., 1983). However, in pilot studies, a clarifier loading rate of 0.62 $m^3/m^2/d$ was achieved (Huck _et al_., 1974). With sludge recycle, loading rates are much higher and reported values for full scale operations ranged from 16 $m^3/m^2/d$ to 100 $m^3/m^2/d$ (EPS, 1985; Kostenbader _et al_., 1970; Lecuyer, 1983).

Polyelectrolyte addition may be used to enhance solid/liquid separation in clarifiers since they cause more rapid settling of the solids and produce a cleaner effluent. Polymer additions increased settling rates from 1.8 to 7.4 m/h and from 0.45 to 2.4 m/h (Scott _et al_., 1975 and Huck _et al_., 1974, 1978). Anionic, low-charged, high molecular weight polymers were found to perform best (Huck _et al_., 1978). From another study on polymer use, Huck _et al_. (1977) could not define an optimum polymer dosage but did relate the dosage to AMW strength as 1.7×10^{-3} x initial iron concentration.

Rapid mix and flocculation energies are important parameters in ensuring effective polymer usage. In pilot plant studies, flocculation was found to be a critical process in obtaining low metal concentrations (Huck et al., 1978). Rapid mixing at 1 250 rpm for 1 sec followed by flocculation at 20 rpm for 25 minutes was found to be optimum. It is important to note that for in-line flocculant addition (polymer), the addition point should be such that adequate mixing is achieved but without severe agitation so that the floc formed will not be broken up.

Effluent Polishing - Advanced (tertiary) treatment may sometimes be required to enhance metal removal. Effluent polishing options include quiescent settling (secondary settling ponds), sand filtration, sulphide precipitation, and water softening. A common effluent polishing stage is settling in secondary or polishing ponds. Canadian operations that send effluent to a polishing pond prior to discharge to a receiving water were described by Kuit et al. (1977), Lecuyer (1983) and Wasserlauf (1985).

Huck et al.,(1978) evaluated several clarifier effluent polishing options during pilot studies: sand filtration, quiescent settling, and sulphide addition. Sand filtration (15 to 25 $m^3/m^2 \cdot h$ hydraulic loading) proved effective in removing particulate metals. For example, concentrations of 0.25 mg/L Pb, 0.36 mg/L Zn, 0.05 mg/L Cu, and 0.28 mg/L Fe were further reduced to 0.09, 0.15, 0.03, and 0.14 mg/L respectively. Quiescent settling (24 h bucket test) effected significant reductions in clarifier overflow metal concentrations (as given above) resulting in a final effluent quality of 0.14 mg/L Pb, 0.23 mg/L Zn, 0.05 mg/L Cu, and 0.14 mg/L Fe. The use of 30 mg/L sulphide (BaS) as a metal scavenger proved ineffectual in the pilot tests.

Sludge Thickening - Sludge thickening can be accomplished using treatment plants that incorporate mechanical clarifiers with thickener rakes, with sludge recycle enhancing the process. Recycling a portion of the sludge with lime through a sludge reactor prior to addition of AMW can lead to much higher solids concentrations. Also mechanical dewatering techniques can also be employed to reduce sludge volumes. These options are discussed later.

A conventional system can be modified to a "High Density Sludge" (HDS) process by adding a second reaction tank (Figure 3) and a return sludge system. The HDS process yielded solids densities of 20 to 35% compared to conventional treatment densities of 1 to 2% (Bosman, 1983; Lecuyer, 1983; EPS, 1985; Kostenbader et al., 1970), thereby significantly minimizing the volume of waste sludge and maximizing the volume of effluent to be discharged. The maximum solids concentration is related to the chemical composition of AMW. The ferrous iron/ferric iron ratio appears to be an important factor in the HDS system with higher sludge solids corresponding to higher ferrous iron concentrations of total iron (Kostenbader et al., 1970). When the percentage of ferrous iron was greater than 70%, then the solids concentration approached 50%. When ferric iron dominated the iron ratio in sludge, sludge densities may be limited to 20% solids (Escher et al., 1983).

SLUDGE DISPOSAL

Sludge Characteristics

Sludge properties depend on the nature of the wastewater, the reagent used for neutralization and the treatment process. These characteristics affect the ability of the sludge to settle, densify and dewater.

Physical Properties - Particle size and shape play a predominant role in dewatering sludges (Knocke et al., 1982). As a particle deviates from a spherical shape, the surface area per unit volume increases, resulting in reduced settleability and decreased dewatering rate. The particles will have more voids and occupy a larger volume. A length to width ratio of about 10-15:1 has been observed for lime neutralization sludge. (Prospectively, the ratio can be reduced by controlling crystal growth or breaking up the rod-shaped particles by mechanical means, with consequent better dewatering).

Sludge viscosity or flowability is also important in designing pumping or pond cleaning systems. A major problem associated with low density sludge is its thixotropic property which causes rat-holing when attempting to pump out ponds.

Sludge dewaterability depends upon such factors as the initial clarifier underflow solids concentration, and the age, temperature, viscosity, compressibility, and chemical composition of the sludge.

Chemical Properties - The chemical composition of sludge varies significantly depending on the quality of wastewater and the neutralization method. Sludge is generally composed of hydrated ferrous or ferric oxides, calcium sulphate (gypsum), along with varying amounts of manganese, copper, zinc and other metals. Table 1 details the chemical composition of eight Noranda Group sludges.

Chemical conditions during precipitation will affect the shape and size of sludge particles. A moderate rate of iron oxidation will result in formation of a blocky crystalline Hydroxy compound, while higher oxidation rates may form amorphous ferric hydroxides (needle-like with poor settleability (Wilmoth et al., 1974)). The conversion of amorphous ferric hydroxide to a crystalline form is slow, illustrating an advantage of recycling. Other factors affecting crystal growth are pH, recycle rate in high density sludge systems and, possibly, age of the sludge. Use of additives such as methylene phosphoric acid to alter the particle morphology has been investigated (McCall et al., 1980), but associated chemical costs may make this prohibitive.

Table I. Chemical Composition of Noranda Group Metal Sludges

Sludge	Pb %	Zn %	Fe %	Cu %	Cd %	As %	Sb %
Waite Amulet	0.010	3.04	20.8	0.31	0.0060	<0.001	0.014
Mines Gallen	0.013	16.9	7.03	0.18	0.083	<0.001	0.013
BMS-M No. 6	0.0066	4.27	7.98	0.25	0.0073	<0.001	0.013
BMS-M No. 12	0.47	3.98	9.52	0.062	0.0054	<0.001	0.013
Hemlo	0.0065	0.014	9.28	0.37	0.0015	<0.001	0.68
Geco	0.0052	3.23	31.8	0.048	0.0060	<0.001	0.014
Remnor	0.048	2.07	13.5	0.62	0.0060	<0.001	0.014
Grey Eagle	0.0046	1.74	4.61	0.89	0.0026	<0.001	0.015

SLUDGE HANDLING

Existing Practices

Low density sludge - Low density sludge (LDS) is produced by simple neutralization without recycle. The sludge is usually discharged into a lagoon. Other disposal methods used by Noranda operations include deposition of the sludge with tailings or discharge into an unused open pit mine.

Lagoons offer the easiest and cheapest methods of sludge dewatering. The most effective lagooning is a series of several small lagoons that allow for easier water removal.

Lagooning of the sludge is usually an interim step for sludge disposal. Excessive volumes of water remain in the 0.1 to 2.0% solids when the lagoon is full. Methods to recommission the pond include pumping the sludge out with a dredge. Other floating pumping systems are available which traverse the length and width of the pond for continuous cleaning.

The main advantages and disadvantages of LDS ponds are:

Advantages	Disadvantages
Low maintenance cost.	Wind or thermal disturbance will resuspend settled flocs.
Low operating cost.	Extensive land use.
Low capital cost.	Large reclamation costs.
Sludge storage facility.	Poor control over discharge.
	Short circuiting occurs readily.
	Not acceptable for final storage.

The LDS systems may also profit from the use of an under-drainage system to promote a thicker sludge. It is envisaged that filled ponds will remain dormant until the sludge is thickened via the under-drainage system, evaporation and freeze-thaw action. The pond can then be cleaned and re-used. This would eliminate the use of continual pond building. This in conjunction with flocculant aids might provide a more cost-effective approach to LDS settling, assuming that a final sludge disposal option is available.

High Density Sludge - High density sludge (HDS) is produced by using clarifiers for solid/liquid separation, and thickeners for sludge thickening. Recycling and addition of chemicals are methods that promote a thicker sludge.

The main advantages and disadvantages of HDS disposal are:

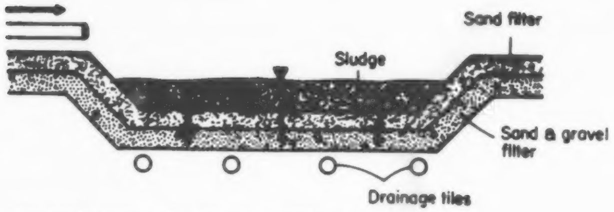
Advantages	Disadvantages
Minimal land use.	Higher capital, operating and maintenance costs.
Reduced volume of sludge for disposal.	Operation requires closer control.
May be suitable for long-term storage.	

Figure 4 illustrates three options currently in use for dewatering HDS. The Noranda Waite Amulet plant near Rouyn, Quebec, utilizes a constructed sand bed underdrainage system. At Noranda's Mine Gallen plant in the same area, sludge is beached on a gravelly bottom bed and the overflow is decanted. Cominco's Sullivan operation in British Columbia uses a natural sand and gravel deposit to dewater its sludge to attain a concentration of 20 to 30% solids.

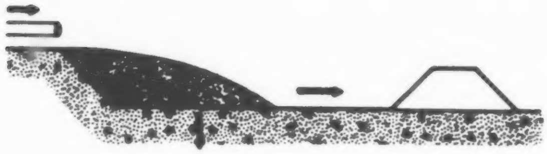
Porous bed drying is a function of sludge and operating variables described below.

Sludge Variables	Operating Variables
Sludge permeability	Sludge application rate
Depth of sludge	Sludge bed construction
Chemical conditioning	Evaporation rate
Sources and type of sludge	Freeze-thaw conditions
Sludge age	
Initial sludge % solids	

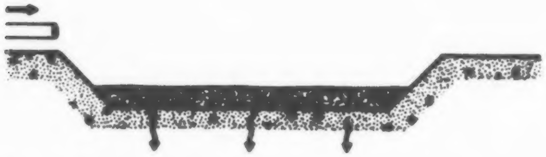
Sludge Pond Optimization - Noranda Research conducted smallscale field trials to investigate the optimum sludge depth, frequency of pond rotation, and to assess freeze/thaw conditioning for the sludge from the Waite Amulet ponds.



a) Gravity drainage filter system (Waite Amulet)



b) Beaching of sludge (Les Mines Gallen)



c) Free drainage sand/gravel areas (Cominco)

Figure 4. Typical Pond Dewatering Methods for High Density Sludge

Columns simulating the gravity sand and gravel "filters" were loaded with 6% solids (existing clarifier underflow) and 11% solids to assess thickening of clarifier underflow. The columns were loaded to different heights and at various time intervals to assess alternative patterns of pond rotation and sludge additions. The sand and gravel filters gave a 16-18% solids for the 6% feed and 19-20% solids for the 11% feed. Sludge depth and addition pattern had little effect.

The 6-11% solids feed, when subjected to a single freeze-thaw cycle, produced a 21-23% solids cake. This increase indicated that freezing liberates more free water from the sludge. Freezing of the sludge also resulted in scouring and reduced blinding at the sludge-sand interface.

Based on these results, Waite Amulet sludge was thickened to about 6-8% solids, discharged to the ponds in rotation, and permitted to drain to about 17% solids. Further dewatering occurs with the freeze-thaw cycle, and the sludge can readily be removed by back-hoe for transportation to a permanent disposal site.

Mechanical Dewatering - More sophisticated dewatering techniques for thickener underflow include the use of vacuum filters, belt presses, pressure filters and solid-bowl centrifuges. Table 2 summarizes the operating parameters, capital, and operating costs for these dewatering methods based on pilot scale studies by Crocker (1982) on a metallurgical refinery sludge. Data are also included from pilot scale sludge dewatering studies that were carried out at Brunswick Mining and Smelting, Mining Division (BMS-M) site #12 (Campbell _et_ _al_., 1975). Other information about AMW sludge dewaterability is available in a "Design Manual" prepared by Akers _et_ _al_., (1973).

Table 2. Thickener Underflow Sludge Dewatering Techniques

	Rotary Drum Vacuum Filter		Belt Filter Press	Pressure Filter		Solid-Bowl Centrifuge	
	Site 1[1]	Site 2[2]	Site 1	Site 1	Site 2	Site 1	Site 2
Feed solids (%)	5.5	7.0	4.5	4.3	7.0	5.0	7.0
Feed Rate (m^3/h)			4.5			1.15	0.683
Filter cake solids (%)	25.0	32-36	33.0	35.0	38-45	20.0	24
Filter yield (kg/m^2/h)	9.8	9-24.5		4.9			
Filtrate solids (%)	0.15	0.1-1.5	0.1	0.15	0.001	0.1	1.5
Solids capture (%)	98		98	97		98	97.5
Cake generation rate (wet t/d)	43.5		33.0	31.1		54.4	
Capital costs ($ 000)[3]	754		640	940		545	
Total sludge costs ($/dry t)[4]	91.50		96.36	97.29		90.82	

Notes:

1 Crocker (1982)

2 Campbell _et al_. (1975)

3 1978 US $

4 Operating costs (including capital depreciation), 1978 US $

From Crocker's work (1982), it was found that the pressure filter and rotary drum vacuum filter gave low filtration rates. The belt filter press should be considered since it produced a dryer filter cake than the solid-bowl centrifuge.

Cake solids of 25% and suspended solids recoveries of 95% were attained on the four pilot-scale dewatering units tested by Campbell *et al*.,(1975). The most important variables for the pilot vacuum filter were the porosity of the filter media and the cycle time. A combination of open weave filter cloth (porosity of 160 cfm/ft^2) and low cycle time gave the maximum yield. The pilot filter press produced the best cake solids and effluent quality. The filter press yield increased with increasing frame thickness (2.54 to 5.08 cm) and operating pressure (7.25 to 21.75 kN/m^2). Optimum performance of the pilot scale basket centrifuge was achieved with a low polymer dosage (0.75 kg/t) and a high feed rate (0.683 m^3/h). However, the use of a basket centrifuge may not be practical due to very short cycle times which resulted from the high sludge feed solids concentration (7%). The solid bowl centrifuge gave best results under conditions of low polymer dosage (0.5 to 1.0 kg/t), deep bowl, and high feed rate.

Sludge Stability

Long Term Storage - Deposition of sludge with tailings is an attractive option since the amount of metals associated with the sludge is insignificant compared to the metals in the tailings mass. However, closure of a tailings area may require an alternate disposal site for sludge from treatment plants which will probably continue operation. Some options are off-site burial, return to process (eg. smelter) or spreading the sludge over a reclaimed tailings area for its neutralization value. Serious consideration must be given to sludge stability to assess its leachability under various storage situations. Studies were undertaken at Noranda Research to assess four leach test procedures on eight sludges from Noranda Group operations to assess their relative stability under alkaline, neutral and acid conditions.

The test protocol was to sample 24 hours after mixing the sludge in an alkaline, neutral or acid condition. Liquid to solid ratios of 4:1, 10:1, and 20:1 were used depending upon the test protocol applied. All tests using distilled water as a media did not result in any significant metal concentration in the leachate. This was due to the alkaline nature of the sludges (pH 9-11). As the media pH was lowered using sulphuric or acetic acid, the concentrations of metals in the leachate increased significantly. A special problem arose when acetic acid was used; Pb dissolved as lead acetate and reported to the leachate at higher values than it did with sulphuric acid.

The testwork illustrated that sludges which maintain an alkaline condition are stable in the short term. Long-term stability under acid rain conditions, or in contact with acid sulphide tailings is not assured, due to the limited buffering capacity of the sludge. In time, the buffering capacity will be used up and the pH will drop allowing the metals to report in the seepage.

Final Sludge Disposal - Final sludge disposal options have received little research to date. AMW sludge contains relatively high concentrations of toxic materials and disposal considerations are critical. Current practice is to use tailings ponds or settling ponds as a means of solid/liquid separation and leave the sludge in the pond as a means of ultimate disposal. Cementation or other fixation processes may provide ultimate stabilization of neutralized sludges.

Sludge disposal methods that can be considered include permanent retention in reclaimed ponds, on site burial or controlled placement into mines, although the last method can have severe drawbacks. The majority of mines in Canada use the tailings impoundment for the final storage of sludge (Scott et al., 1975). Underground disposal is not used in Canada, although it has been reported to be a practice in the USA (Escher et al., 1983).

Sludge usually accumulates in the ponds over the life of the mine (10 to 20 years). The ponds will then require covering or other stabilization methods to prevent future problems. Impoundments require large land areas and planned methods of handling surface runoffs and controlling drainages. The main problem with disposal of AMW sludge in sludge settling ponds is the time necessary for drying before covering or revegetating.

Sludge Utilization - Sludges can be sent to smelters for fixation in the slag as an alternative disposal method when available. A number of examples are presented by Wasserlauf (1985) and EPS (1985). This method reduces or eliminates the problem of AMW sludge storage. Although currently not used for metal recovery, the potential for economical recovery of metals, such as zinc, may make this disposal method attractive.

Some other possible uses for AMW sludge are described by Simonyi et al. Sludge applied at 3% solids at pilot scale as an amendment to strip soil samples improved plant germination and survival. Higher sludge additions were better, apparently having better pH neutralizing (increasing) power. However no leachate data nor data on the uptake of metals by plants were provided. A second option was to use dried, powdered sludge as a rock dust substitute for explosion control in underground coal mining operations, although this is likely an expensive alternative since sludge must be fully dried and pulverized. Thirdly, AMW sludge may be added to cement in amounts of 5% or less with no net effect on strength, but this addition is of little value other than as a method of sludge disposal.

TREATMENT PLANT DESIGN AND COSTS

Currently full scale treatment systems in Canada utilize lime neutralization for AMW treatment. Table 3 presents a summary of process data, treatment effectiveness of existing facilities and capital and operating costs. A wide range of operating conditions have been presented which are being used to treat a variety of wastewaters with a broad range of characteristics. Effective treatment has been accomplished by each of the different facilities, with metal concentrations in the effluents generally much less than 1 mg/L.

Most operations have a treatment system with a mechanical clarifier. Lime has been used in all cases to provide a reaction pH greater than 9.5. In general, the retention time in the reactor is at least 20 minutes. Air is applied in at least 50% of the treatment schemes at a wide range of rates. Polymer dosages from 0.65 to 15 mg/L are being applied to most clarifiers. Very little data were available about sludge recycle but in the one case where information was available, sludge solids of 20% were achieved compared to solids concentrations of 1 to 4% when sludge recycle was not employed.

Another summary of treatment facilities for metal mines in Canada and the liquid effluent quality produced by these facilities is presented by EPS (1985). A design manual, prepared for the US EPA (Escher et al., 1983), assesses the selection of processes, equipment, and procedures in

Table 3. Summary of Full Scale Acid Mine Water Treatment Plants

Treatment Plant Operating Conditions

SITE	FLOW (m^3/d)	WW DESC	LIME DOSE (kg/m^3)	HDS RATE (%)	REACT RT (min)	REACT pH	AIR RATE (m^3/min)	POLYMER DOSE (mg/L)	SETTLER TYPE	SETTLER OVERFLOW RATE (m^3/m^2/d)	SLUDGE SOLIDS (%)	FINAL SLUDGE DISPOSAL	COSTS (YEAR) CAP($000)	COSTS (YEAR) O&M($/$m^3$)
1	5500	leachate	7.4	0	20	9.5	N/A	N/A	pond	20	3	mine	total=400 (cap,O&M)	
2	2600	AMW	1.8	50	45	9.5	N/A	N/A	clarifier	N/A	20	basin	1000 (76)	0.42 (76)
3	1300	AMW	N/A	0	N/A	N/A	0	?	clarifier	28.5			110 (74)	0.15 (74)
4	2160	AMW	6.4	0	50	9	4.5	15	clarifier	9.2		tailings	800 (84)	0.70 (84)
5	4080	leachate	3.76	0	9	<10	0	0	open pit			open pit	300 (84)	0.41 (84)
6	4320	leachate	1.1	N/A	45	9.5	12	5	clarifier	18	4	pond		0.43 (84)
7	6600	TPD	0.04	N/A	20	9.8	0	0.65	clarifier	20	4	smelter	1100 (84)	0.015 (84)
8	4320	mixture	1.5	0	25	9.3	11	5	clarifier	10.4	2.1	tailings	1300 (84)	0.19 (84)
9	57000	leachate	N/A	?	?	9-10	0	?	ponds		1	injection		
10	270000	mixture	N/A	?	N/A	7-9.9	?	N/A	clarifier	100		tailings		
11	36000	mixture	N/A	N/A	N/A	9.5-11.4	?	?	clarifier	N/A		smelter		
12	26000	AMW	N/A	N/A	N/A	9.5	N/A	?	clarifier	N/A	20-30	pond	10000 (79)	0.10 (83)

Effectiveness of Treatment: Comparison of Influent (I) and Effluent (E) Concentrations (mg/L)

SITE	pH		Fe		Cu		Pb		Ni		Zn	
	I	E	I	E	I	E	I	E	I	E	I	E
1	3.5	9.8	2100	1.6	55	0.24					580	0.58
2	3.5	6.5-9.0	865	0.75	0.26	0.04					530	0.45
3			300	0.2	100	0.01					100	0.09
4	3	9	1700	<4	12	<0.1	7	0.25			1200	<4
5	<2.8	<10	800	<4	20	0.2	1	0.7			500	<2.5
6	2.8	9.5	254	1.4	3.5	0.02	0.05	<0.05			20.7	0.2
7	5.8	9.8	1.5	0.05	0.06	0.01					10	0.5
8	3.9	9.3	460	1.1	3.4	0.1	<0.05	<0.01			120	0.4
9	4.4	9-10	2	0.08	6	0.02	0.02	<0.02			40	0.3
10	5.8	7-9.9	15.4	0.6-1.3	0.6	<0.03			9.2	0.03-0.9		
11	4.3	9.5-11.4	35.6	0.3-3.4	6.7	0.02-0.6			19.8	0.2-1.3		
12	4.5	9.5	250	<1.0	0.15	0.05	5	<0.1			22-75	<0.2

Legend:

AMW = acid mine water
"leachate" from waste rock runoff, etc.
N/A = process used but data not available
Sites:
1 = Wedge Mine (closed) (Kuit et al., 1977)
2 = Les Mines Gallen (Lecuyer, 1983)
3 = Chester Mine (Scott et al., 1975)
4 = BMS-N #12 (Wasserlauf, 1985)
5 = BMS-N #6 (Wasserlauf, 1985)
6 = Waite-Amulet (Wasserlauf, 1985)

TPD = tailings pond decant
"mixture" = any combination of above
? = no information available about process

7 = Mattabi (Wasserlauf, 1985)
8 = GECO (Wasserlauf, 1985)
9 = Westmin (EPS, 1985)
10 = Inco Copper Cliff (EPS, 1985)
11 = Inco Nolin Creek (EPS, 1985)
12 = Cominco (EPS, 1985)

treating AMW. Design details are provided of chemical storage and feeding systems, mixing, aeration and neutralization facilities, sedimentation facilities, sludge handling, dewatering and disposal facilities. A description of process control equipment and estimated costs for unit processes are included.

SOME AREAS FOR FUTURE WORK

Based on the above review of current practices and technology, the following areas appear to warrant further study:

- improvement of the settling, densifying and dewatering characteristics of the sludge.
- demonstration of in-situ dewatering of stored low density sludge.
- reduction of the leachability of the sludge.
- economical recovery of the metal values in the sludge.
- effective scale control without additional treatment or chemical additives.
- assessment of appropriate sludge recycle rates.

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ECOLOGICAL ENGINEERING: BIOLOGICAL AND GEOCHEMICAL ASPECTS
Phase I Experiments

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ABSTRACT

Perpetual operation of acid mine drainage treatment facilities after mine closure imposes economic burdens and environmental liabilities difficult to overcome. The main goal of Ecological Engineering research is the development of methods which facilitate the establishment of vegetation cover types, serving as self-maintaining acid drainage treatment systems. Objectives and results of experiments on tailings and in the laboratory are summarized for 5 aspects relevant to the development of close-out scenarios for acid generating tailings sites in temperate climates.

1) On dry surface areas with sparse vegetation covers terrestrial moss covers curtail infiltration of precipitation. Vascular plant covers investigated had no amelioration effect on the underlaying tailings. Introduction and promotion of terrestrial mosses requires further research.

2) Wetlands established in waterlogged, submerged sections and seepages reduce water infiltration, acid generation and remove metals. Hand-transplanted cattails showed a 200 % increase in number after one year. Hydroponic methods tested to date indicated little promise. Seed germination of some wetland plants was reported on neutralized pyrrhotite. Iron removal in seepages was found to be catalyzed by the surface provided by organic matter. Two species of aquatic mosses can serve as biological polishing agents where flow rates are low.

3) The precipitates in settling and polishing ponds from treatment plant operation have to be maintained alkaline. Some species of Charophytes, attached macroscopic algae, have been tested extensively for these conditions and appear suitable.

4) Abundance values of O^{18} for the product sulfate can be used to estimate the proportions of sulfate oxygen derived from molecular (atmospheric or dissolved) oxygen and from meteoric water (rain and snowmelt) during pyrite oxidation. This may provide a suitable method to determine oxidation and acid generating zones within the tailings mass and its hydrogeochemistry.

5) Freezing gauges and thermister cable measurements taken inside the tailings mass indicate that frost penetration is less in the tailings than in soil, suggesting that a reduction in the rate of oxidation in winter may be less than anticipated.

ACKNOWLEDGEMENTS

The financial support of BP-Resources,Selco Division; Denison Mines Ltd.; Falconbridge Ltd. Sudbury operations; Inco Ltd. Sudbury operations; and Rio Algom Ltd. is gratefully acknowledged. The financial support of the Department of Energy Mines and Resources Canada, CANMET - Biotechnology , and the National Research Council who furnished several contracts and grants, was essential to this research and is acknowledged and appreciated.

INTRODUCTION

One of the most challenging aspects of reclamation is the maintenance of stable and enduring environmental protection against water quality degradation from acid generation by sulfide bearing mining wastes in temperate climates.

In Canada, it has been estimated that at least 9000 ha are covered by sulfide tailings, mainly located in the provinces of New Brunswick, Quebec, Ontario, Manitoba, British Columbia and the Territories (RATS,1984). Given that acidic effluents are continuously being released from these waste sites throughout the country long-term environmental protection is required. Currently operation of acid mine drainage treatment facilities, long after mine closure, impose economic burdens, through continuous demand of maintenance, which are difficult to face for mining companies and governments to face. Self maintaining ecological systems capable of treating acidic water and progressively curtailing the acid generation process from the wastes, could provide an economically sound method for long term environmental protection.

Many ecological studies of plant colonization patterns have indicated that on waste sites, it is likely that species richness and the extent of the colonization will not change for decades or millennia (Kimmerer, 1981; Bauer, 1973; and Morrison & Yarranton, 1973). Ecological projections indicate that improvements due to natural recovery are unrealistic, despite some initial evidence of primary natural colonization of wastes sites. The establishment of species tolerant to the chemical and physical characteristics of the waste sites, appears to be a random process of chance. Self-sustaining covers on waste areas are often limited to localized areas of the site where conditions happen to be favorable for germination and establishment (Kalin and Caza, 1983). Therefore methods have to be developed to provide suitable conditions, unilaterally favorable for self-sustaining vegetation covers comprised of those highly tolerant species. These covers would be the mainstay of self-maintaining mine drainage treatment systems.

The general concepts for the development of Ecological Engineering and Biological Polishing methodologies and procedures for treatment systems have been formulated (Kalin, 1986a and Kalin and Smith, 1986). The target of Ecological Engineering research is the development of methods facilitating the establishment of self-maintaining vegetation covers for the different conditions encountered at acid generating waste management areas. These methodologies can then be applied to promote vegetation covers and wetland communities which would function as biofiltration systems.

Biofiltration and its applications to the treatment of mining effluents is not novel and has been reviewed by Barth (1986). He suggests that the use of processes associated with biofiltration "may offer a low cost, highly effective, maintenance-free approach to treating some types of tailings seepage." Although several examples of working biofiltration systems are presented, he cautions that until the knowledge on biofiltration by wetlands increases, widescale application of biofiltration is premature.

Summarized in this paper is on-site experimental research and concurrent laboratory studies initiated on two tailings sites. The objectives and results of the ongoing investigations are given. Detailed descriptions of the methods employed are not presented.

Technical and scientific information further describing the processes at work, together with on-site applications of the methods developed will demonstrate in the near future Ecological Engineering measures which can provide an acceptable alternative to the presently available treatment options for acid mine drainage.

TAILINGS SITE DESCRIPTION

One of the experimental sites is a pyrrhotite covered tailings area in Sudbury (Central Ontario), where milling operations have ceased. Neutralized mine water covers part of the site. This pond receives acid water pulses during precipitation events from the pyrrhotite-covered area. A seepage, emerging at the foot of one of a tailings dams, is used as a study area to determine those factors controlling Biological Polishing processes, i.e removal of metals.

A second experimental site is a waste management area located in the Red Lake area (Northern Ontario). Operation of the zinc and copper concentrator ceased 4 year ago, after a 10 year life span of the mine. Reclamation of tailings, containing 3 % pyrrhotite and 25 to 30 % pyrite, was implemented recently. During operation of the mine a small lake adjacent to the tailings became acidified. The original decant pond containing sludges from water treatment and fines, receives drainage and run-off from the elevated tailings area. A complete close-out of this site is being implemented, using Ecological Engineering measures.

REQUIREMENTS FOR ECOLOGICAL TREATMENT SYSTEMS

Various surface and seepage characteristics, as well as specific behavioral aspects of the tailings mass, are primary considerations during the development of a close-out scenario for a waste management area. The tailings mass characteristics are related to the pattern of tailings deposition within the geomorphological and hydrological setting of the site.

The surface characteristics can generally be classified into three groups: 1) vegetated reclaimed surfaces with vegetation free acidic areas; 2) waterlogged, submerged sections and seepages from dams; and 3) settling and polishing ponds remaining from the treatment plant operation. Behavioral aspects of the tailings mass relevant in the long term are; 4) oxidation and acid generating zones within the tailings mass and the hydrogeochemical characteristics of the tailings; 5) the seasonal variations in the oxidation process.

Fulfillment of the overall objectives of a self-maintaining close-out require development of specific ecological systems for the three surface types (1 through 3 above) while decisions concerning the locations of these systems within the waste management area have to be based upon the behavior of the acid generating tailings mass (4 and 5 above). The specific objectives addressing the three surface types of the waste management area are outlined briefly.

1) Dry areas with sparse or no vegetation cover: Vegetation covers are required which would reduce infiltration of precipitation. Field observations by M.Kalin on several acid generating tailings sites consistently indicate that covers

of terrestrial moss might serve this role. This was also suggested by Barth (1986). A moss cover has no roots, compared to vascular plants which penetrate the tailings and thereby increasing porosity. Terrestrial mosses have shallow rhizoid mats which serve to anchor the moss and provide moisture storage, particularly on tailings surfaces prone to severe seasonal droughts. Thus a fraction of the precipitation is retained in the moss and rhizome layer not infiltrating the tailings.

2) Waterlogged, submerged sections and seepage areas: Wetland vegetation covers increase transpiration and thereby counteract infiltration. That wetlands are effective means of removing water is suggested by water economy evaluations of plant communities by Larcher (1980). He reports that, in a wet meadow, 135 % of total precipitation per year is transpired, compared to that of a dry grassland, where only 30 % is transpired. A wetland would result in a net export of water from the site.

Furthermore wetlands will promote the accumulation of organic matter. This layer will ultimately provide reducing conditions, which would cause some decrease of sulfate in the water. The amount of acidity increase would therefore be less than in the absence of a wetland cover. Reducing conditions curtail the activity of Thiobacilli. A schematic of the wetland development and its ultimate self-maintaining state is outlined in Fig.1. Colonization of the wetland by appropriate algal and aquatic moss species should remove metals from the water, as some species have demonstrated a capacity for biological polishing.

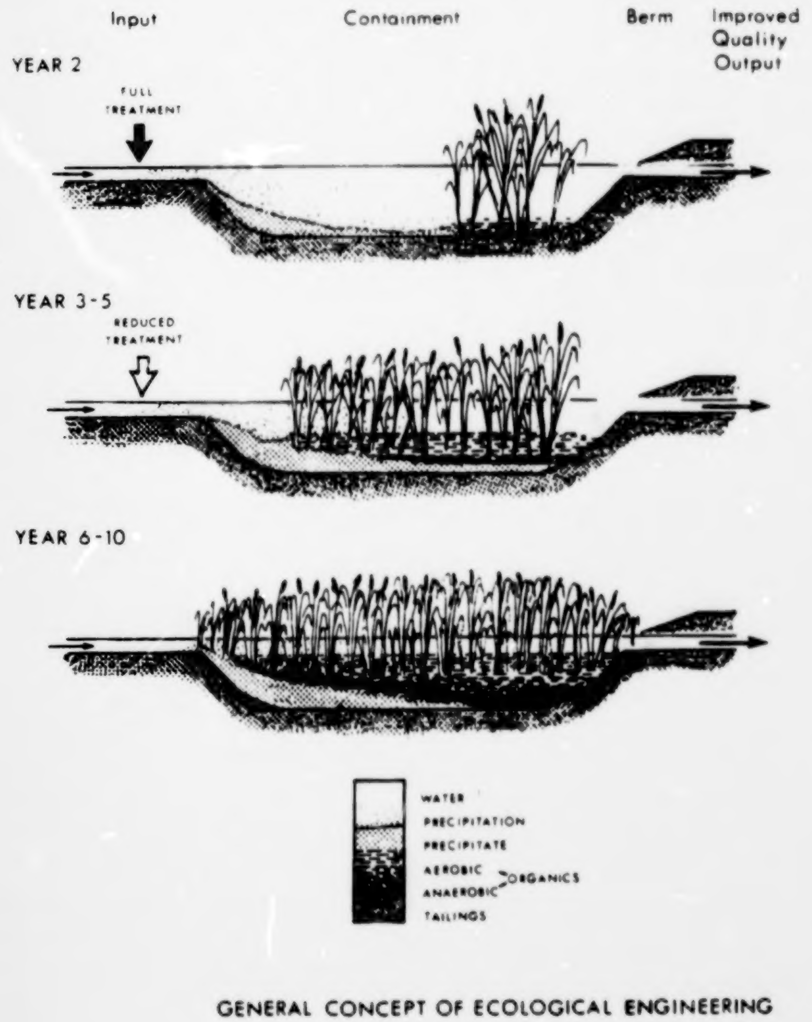
Input
Containment
Berm
Improved Quality Output
YEAR 2
FULL TREATMENT
YEAR 3-5
REDUCED TREATMENT
YEAR 6-10
WATER
PRECIPITATION
PRECIPITATE
AEROBIC
ANAEROBIC
ORGANICS
TAILINGS
GENERAL CONCEPT OF ECOLOGICAL ENGINEERING

Fig 1: Schematic development of wetlands on waste sites.

3) Settling and polishing ponds from previous treatment: These areas are often very alkaline and filled with precipitates and sludges generated by the treatment system to be decommissioned. These areas need to be maintained so that potential acidification with remobilization of metals does not occur. Biological polishing agents are required which are tolerant to the alkaline conditions and capable of actively maintaining water at neutrality.

The requirements outlined above have to be connected to the behavioral aspects of the tailings mass and the rationale for their respective importance is outlined below.

4) Acid generation within the tailings mass: The generation of sulfuric acid during oxidation of pyrite and pyrrhotite in tailings is generally assumed to be the result of microbial action under aerobic conditions. When acidification proceeds to less than pH 3, the ferric iron remains in solution and acts as an oxidizing agent. Acid generation can be expected to take place under saturated as well as under aerated conditions and also within the relatively narrow zone of alternating wetting and drying (water-level fluctuations). It is reasonable to assume that not all locations in the tailings mass are in the same condition with respect to water saturation and oxygen availability. It follows that parts of the tailings mass may not generate acid and/or that the rates will differ in various locations where acid generation does occur.

The conditions under which the oxidation proceeds determine the relative proportions of the sulfate oxygen that are derived from atmospheric (or dissolved) oxygen and from water molecules (Taylor et al. 1984a, 1984b; van Everdingen and Krouse 1985). As the relative abundance of O^{18} in atmospheric oxygen and in meteoric water (rain and snowmelt) are very different, particularly at higher latitudes, O^{18} abundance values for the product sulfate can be used to estimate the proportion of sulfate oxygen derived from the two sources. This in turn would give an indication of the conditions under which the oxidation process operates. Determination of the oxygen source would greatly assist in the placement of the surface treatments within the waste management area.

5) Seasonal changes in acid generation: Temperature fluctuations and freezing of the tailings mass has not been documented to date. Seasonal fluctuations in temperature of the tailings mass are expected and would govern variations in the rate of acid generation. It is essential to relate seasonal differences of acid generation within a waste management area to the growth cycles of the wetlands and the other biological systems.

A reasonable assumption would be that reduced biological activity during the winter month would be compensated for by reduced acid generation rates. However in principle, the oxidation process in the tailings mass is associated with the generation of some heat which would tend to delay or even prevent freezing of the tailings. In addition, freezing point depressions can be expected due to the generally high dissolved solids content of the pore water in the tailings. Wetland design may have to incorporate means by which acid generated during the winter months can be accommodated during spring freshet.

EXPERIMENTAL RESULTS AND ONGOING INVESTIGATIONS

Work relevant to each of the areas for which the requirement for ecological treatment systems have been outlined is presented.

1) Dry areas with sparse or no vegetation cover: A vegetation cover after 4 years of reclamation was evaluated in detail with respect to interactions of the cover with the underlying tailings. Three types of grass covers could be differentiated (sparse, background and dense) which grow on a 30 to 40 cm thick fill layer. The thickness of the fill layer, depth to oxidation layer, depth to tailings, depth of root penetration, and thickness of organic layer were measured. In addition, measurements of pH on the surface, in the fill and in the tailings were recorded. Temperature was determined on the surface and in the tailings. All measurements were performed in pits dug to the depth of tailings in the various surface cover types.

An analysis of the results indicated that there was no relationship between the vegetation type, the thickness of the fill and the depth at which the tailings were acidic (pH <3.5). The tailings were consistently acidic 2 to 3 cm below the fill, and only in areas with a gravel cover alone were the tailings acidic at a shallow depth (gravel cover depth 10 cm as compared to 30 to 40 cm of fill). The temperature on the surface of the vegetation was lowest in the dense grass cover (19 ^{o}C) as compared the other cover types (23 to 28^{o}C). However the temperature in the tailings below did not vary between cover types (11 to 15^{o}C).

The vascular plant species composition was determined based on collection of species most frequently occurring on the tailings after four years of reclamation. The composition was typical for waste lands, consisting of sedges (Carex aquatilis c.f.) and grasses such as Wild rye, Fuzzy wild rye, Orchard grass and Quack grass (Elymus sp., Agropyron repens c.f., Dactylus glomerata), Foxtail barley (Hordeum jubatum), Smooth brome grass (Bromus inermis), Woolgrass or Bog cotton (Scirpus cyerinus), Birdsfoot trefoil and Alsike clover (Lotus coniculatus and Trifolium hybridum).

Annual above-ground biomass production for the sparse grass cover was 15 g/m^2, and 53 to 57 g/m^2 for the background and dense cover respectively. A one time application of fertilizer (21-7-7) of 6.25 g/m^2 in the beginning of the growth season resulted in a net increase of biomass of 84 to 110

g/m^2. However, fertilization of the terrestrial moss cover applied at the same rates did not result in any noticeable changes in the cover extent or its growth.

Field and greenhouse experiments addressed methods of introduction of various terrestrial moss species to the pyrrhotite surface. In the field, two types of transplanting techniques were tested. Clumps of mine slimes bearing protonemata of Leptobryum pyrifome were dug into the surface of loose pyrrhotite. The innocculated mine slimes were slurried by circular motion during rain into the broken pyrrhotite. The pyrrhotite was amended with either sand or sandy topsoil.

The slurry technique indicated some promise in conjunction with sand amended pyrrhotite. The greenhouse experiments produced similar results with respect to both introduction and amendment techniques (Kalin,1985).

2) Water covered, waterlogged and seepage areas: Experiments were implemented to ascertain some of the factor(s) which control cattail (Typha latifolia and glauca) growth and stand expansion on alkaline mine slimes covering pyrrhotite tailings. It was found that the ratio of the number of juvenile individuals to the number of mature individuals around the cattail stand perimeter was lower in stands where the compaction of the waste was high and the rhizomes were buried deeper. A rhizome depth of greater than 10 cm had

lower numbers of juvenile plants (Kalin & Buggeln,1986a).

Transplanting methods for introduction of cattails in areas free of any vegetation were tested. Mature cattails from various sources (stands in gravel pits, in dense cattail meadows, stands growing in tailings, stands growing in mine slimes) were transplanted by hand (individuals or clumps) and mechanically by a front end loader. Hydroponic means of transplanting were tested, where root stocks were suspended in fishing net and mature cattails fastened in rafts.

The results from the hand transplanted cattails in the mine slimes after overwintering and growth for one season are summarized in Table 1. A total of 8 stands were established, however the markings on the plants did not survive the overwintering and data could not be obtained (Kalin, 1986a). The data in Table 1 pertain to the marked plants.

Table 1:
Handtransplanted cattail survival and growth 1985/1986

Stand #	Number transp.	Number Overwintered	Total Plants June / Sept.	% Increase over 1 season
6	10	8	18 / 27	238
7	10	9	21 / 27	200
8	10	8	14 / 21	162
10	10	7	16 / 21	200

Hand and mechanical transplanting was tested in acid water with pH values ranging over one growing season from 2.9 to 5.7, with a seasonal mean zinc concentration of 221 ±79 mg/l and mean copper concentrations of 21 ± 36 mg/l. Such high concentrations of metals, though not unusual for waste management areas, present extremely unfavorable growth conditions.

Two months following transplanting, there was little difference between the two methods. The number of new shoots produced by mechanical and hand-transplanted populations had increase of 32 to 35%. The control (natural environment) populations transplanted by the same methods increased by 36 and 42% respectively. Populations transplanted to mine slimes at a shallow depth increased on average 100% in within two months.

Hydroponic methods yielded less encouraging results in both acid (pH 2.5) and alkaline (pH 8 - 10) although shoots did develop initially. None of the hydroponics survived the overwintering from the 1985 experiments, since both alkaline and acidic test sites had dried up by the end of the growing season. In experiments carried out in 1986, the control populations in fishing nets initiated shoots after two months and racks showed some survival in alkaline water.

Seeding would clearly be one of the most desirable methods of introducing wetland vegetation. Seeds collected from a variety of semi-aquatic plants growing in a polishing pond were tested for germination. The seeds were subjected to various pre-germination treatments (freezing, desiccation and storage at room temperature). Seeds from an aquatic grass, a sedge and cattails stored at room temperature showed best germination, whereas seeds from sedges and a rush failed to germinate, regardless of the pre-treatment.

Seeds which showed positive germination after pre-treatment were further tested on pyrrhotite, amended with either dewatered sludge, with Ca $(OH)_2$ (pH 3.0, 7.0 or 10.0) or left unamended (pH 2.3 to 2.5). No seeds germinated on sewage-amended, pH 3.0 amended or unamended pyrrhotite. Excellent germination (50 to 100 %) occurred with the grass and a sedge on pH 7.0 to 10.0 amended pyrrhotite. Cattail seeds, collected from different habitats (acid, alkaline or natural environment) however, showed varying degrees of germination rates (Kalin, 1986c).

A seepage creek was analyzed in detail for its chemical, physical and biological characteristics. Nine stations at 50 m intervals along the creek were monitored monthly from June to August, and at irregular intervals for a complete year. Based on the data collected, the following processes can be described.

Organic matter (living and dead) in the seepage creek provide a catalytic surface for the precipitation of iron hydroxide from the water. Steady iron removal occurs over the upper 250 m of the creek. The hydraulic retention time of the creek increases as the water enters a broad expanse (=pond) mainly colonized by (Calamagrostis spp). A 6.5 fold reduction in total soluble iron occurs across this area along with a concomitant drop in pH of 2 full units (Kalin,1986b).

In the grass covered, broad expanse, estimates of organic matter and iron precipitate were made. The results indicated that in dense or medium dense areas (live grass cover), an average of 8 g (air dry) iron precipitate was associated with 1 g (air dry) dead organic matter. In areas with sparse growth, 1 g of dead organic matter was associated with 22 g of precipitate. Dead organic matter estimates for the densely and medium growing areas were 3.8 to 4.8 kg/m^2 and for the sparse 1.9 kg/m^2. These values suggest that there may be an upper limit of precipitate at which continuous growth of grass might be inhibited.

Laboratory studies of the cation exchange and binding capacities for metals of acidophillic aquatic moss species _Drepanocladus_ _fluitans_ and _Spagnum_ _riparium_ were carried out. These species are tolerant to high sulfate concentrations, evident from their colonizing inactive tailings areas with pH 2.5 to 3.0.

Growth rates of shoot for both species were 3 cm/ month. The biomass increase differed between the species, due to differences in morphology and growth habit. _D._ _fluitans_ produced 2.1 g/m^2/month dry weight as compared to _S.riparium_ which produced 17.2 g/m^2 /month. Under field conditions, it can be expected that the respective dry matter production will range from 20 to 170 g/m^2/month in the growth season.

The cation exchange experiments indicated that 1 g (dry weight) of moss can remove 3 ug of nickel from 100 mLs of seepage water (pH 3 to 4) with concentrations of 0.1 mg/L of nickel. Similar amounts of removal were observed with Cu and Mg. Increasing the iron concentrations in the seepage water (up to 36 mg/L) inhibited the metal uptake. (Kalin & Buggeln, 1986b).

3) Settling and polishing ponds from previous treatment: Some species of Charophytes, a group of macrophytic attached algae, have been tested in alkaline waters of settling and polishing ponds of eight uranium sites and in ponds from two nickel operations (Kalin,1985). Growth tests in the field and the laboratory indicate, that the algae concentrate uranium, radium 226, nickel, copper and zinc under alkaline conditions (Kalin & Smith,1986). The algae appear to provide pH stabilization due to accumulation of calcium carbonate on its surface. Physiological studies (Lucas,1985) indicate that this algal group can utilize bicarbonate at pH 8.5 to 10.0 during photosynthesis after depleting carbon dioxide. This results in maintenance of neutral water, if populations can be developed with high densities.

Ponded areas of tailings sites of several abandoned gold mines have been colonized by Charophytes to varying degrees. The growth upon transfer to settling ponds from some of these populations was considerably better than from population previous tested. Factors controlling population densities and growth in the field are presently being investigated. Experiments on inactive tailings ponds are being developed to address stimulation of the growth of the algae.

4)Acid generation within the tailings mass: The conditions under which the oxidation process operates within the tailings mass have been addressed through an analysis of stable isotope abundance ratios. In water collected from piezometers installed in, below and outside the tailings mass, the following ratios have been determined: ^{34}S and ^{18}O in dissolved sulfate, and ^{18}O and ^{2}H in water molecules. The isotope ^{34}S in the metal sulfides contained in the tailings has also been determined.

The $^{34}S/^{32}S$ values for the sulfates and sulfides should indicate whether any significant fractionation of sulfur isotopes occurs during the oxidation process. The $^{18}O/^{16}O$ values for the sulfates should reflect the proportion of water oxygen incorporated in the sulfates. Comparison of the $^{2}H/H$ and ^{18}O / ^{16}O values for the waters should give an indication of the source of the water (rain and snowmelt).

The dominant reaction process in the oxidation of metal sulfides in the tailings should then be determined from a comparison of the relationship between the $^{18}O/^{16}O$ values for sulfates and water in the piezometer samples with similar data for:

a dissolved marine sulfates (van Everdingen et al. 1982)
b acidic spring waters (van Everdingen et al. 1985);
c acid mine waters (Taylor et al. 1984a); and
d pyrite oxidation experiments (Taylor et al. 1984b).

Initial isotope data indicate that: 1) The water that is found in the tailings is derived mainly from rainfall not snowmelt; 2) 43 to 83 % of the oxygen incorporated in dissolved sulfates in the tailings is derived from H_2O molecules, indicating that, 3) 35 to 80 % of the dissolved sulfate is produced by a reaction not involving free (or dissolved) oxygen; this in turn suggests that oxidation of iron sulfides in the tailings investigated is taking place under both aerobic and anaerobic conditions, with the probable involvement of *T. ferroxidans* or similar bacteria.

The geochemical analysis of waters from piezometers located in tailings indicate extremely large ranges in chemical composition. For example, the sulfate concentrations in the same tailings deposit ranged between locations from 17,000 to 132 mg/l with iron concentration in the same two locations ranging from 9860 to 16 mg/l. These ranges cover 15 piezometers located in the tailings mass. For water from 17 piezometers below the tailings mass located either in sand or bedrock similarly large ranges were encountered.

The concentrations of metals Zn, Ni, Hg, Pb and Co in the tailings piezometer water varied by two orders of magnitude over all tailings piezometer locations, whereas the elements Al, As, Ca and Na varied by only one order of magnitude.

From the water collected from 28 piezometers located outside the tailings area the geochemical characteristics suggest that the contaminant plume moves at different rates in various directions.

5) Seasonal changes in acid generation: Seasonal temperature changes and freezing behavior of tailings are being determined using instruments inside and outside the tailings areas. Freezing gauges after Banner and van Everdingen (1979) were installed in August 1986. Measurements of temperature with a thermister cable installed together with the freezing gauge inside the tailings are taken weekly.

The preliminary data from the freezing gauges and the thermistor cable suggest that frost penetration in the tailings is minimal. Initial cooling to minus 1.3 °C at a depth of 23 cm was reversed when snow cover was established; on January 4th 1987 subsurface temperatures ranged from near zero at ground surface to plus 8.6 °C at 2.43 m depth. These data likely reflect the generation of heat at some depth in the tailings.

CONCLUSIONS

The conclusions drawn from the result presented are discussed in context of the requirements to be achieved for each of the 5 components relevant for the development of Ecological Engineering treatment systems.

1) For the dry areas, with sparse or no vegetation cover, terrestrial moss establishment and growth would be desirable. Although some methods to introduce acid tolerant moss indicate promise, further extensive work is required. Different densities of vascular vegetation covers on tailings appear insignificant with respect to curtailment of the acid generation underneath the covers. Four years after reclamation, the species composition of the cover has changed. The area has been colonized by indigenous species, representative of waste sites. Fertilization measures did increase the biomass of vascular plants while the growth of terrestrial moss was not affected. Greenhouse and field tests of introduction methods for terrestrial mosses indicate, that modification of physical conditions (moisture, structure of the surface etc.) will likely promote moss development.

2) Wetland development methodology requires a better understanding of the conditions under which isolated pockets of the waste sites are colonized, particularly by cattails. One factor inhibiting the expansion of stands is burial of rhizomes under compacted material deeper then 10 cm.

Direct germination and establishment of seeds by some wetland species does occur in alkaline conditions (on mine slimes) or on neutralized pyrrhotite. The results suggest that seedling establishment in neutralized or alkaline is the origin of the isolated populations on waste sites.

Mature cattails transplanted by hand or mechanically survived and continued to grow in both alkaline and acidic conditions. Mechanical transplant methods are restricted to areas where equipment access is possible, however growth was comparable to hand transplanting. The hydroponic transplanting methods tested require further development.

Investigations of seepages indicate that the initial iron removal occurring as the water emerges under neutral conditions is due to a catalytic process, related mainly to the surface area of biomass. Active uptake by algae, although occurring, is secondary; the precipitation processes dominate. Therefore the continued growth of wetland species is of importance providing surface area for iron removal. Biological polishing of seepages could be achieved in two steps. At the head of the seepage a neutral ecosystem is required to remove iron, followed by acid-tolerant system for metal binding.

The growth rates and cation exchange capacity of two species of acid-tolerant aquatic mosses indicate high metal removal could not be achieved by these two characteristics alone.

However in ecosystems with low flow rates these species' contribution to metal removal might be significant through their provision of nucleation sites for adsorption processes.

3) Field and laboratory results indicate that the Chara Process has the potential to provide a buffering capacity for the water and fix alkaline precipitates in settling and polishing ponds.

4) A working hypothesis for the determination of the oxidation process and its oxygen source derived from isotopic ratios will be tested in the future. If the trends indicated so far persist, that is, water in the tailings originates mainly from rainfall and not from snowmelt, and that oxidation of iron sulfides occurs both aerobically and anaerobically, then realistic, site-specific predictions of active areas of acid generation within the tailings mass will be possible.

The geochemical data from an extensive sampling program of piezometer water in various locations of the same tailings mass and outside the tailings impoundment support the hypothesis that different oxidation processes are occurring in the tailings.

5) The temperature behavior of the tailings mass suggests that, at best, a slight cooling can be expected in the win-

ter. Since the frost penetration is considerably smaller than expected, only an insignificant reduction in the acid generation rate can be expected.

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A

COAL REFUSE CASE

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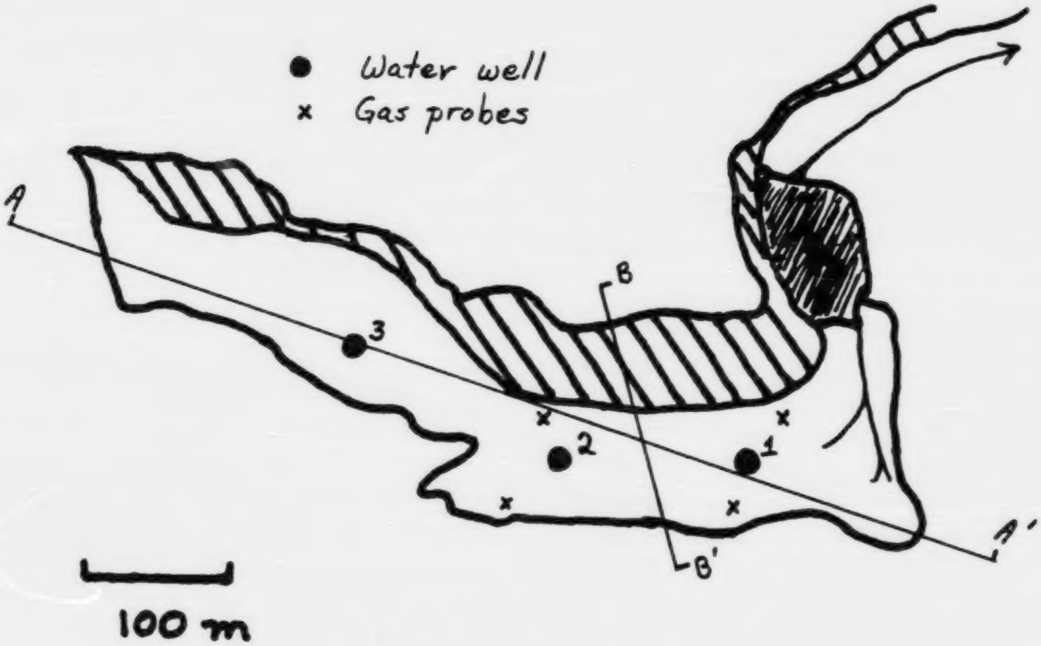
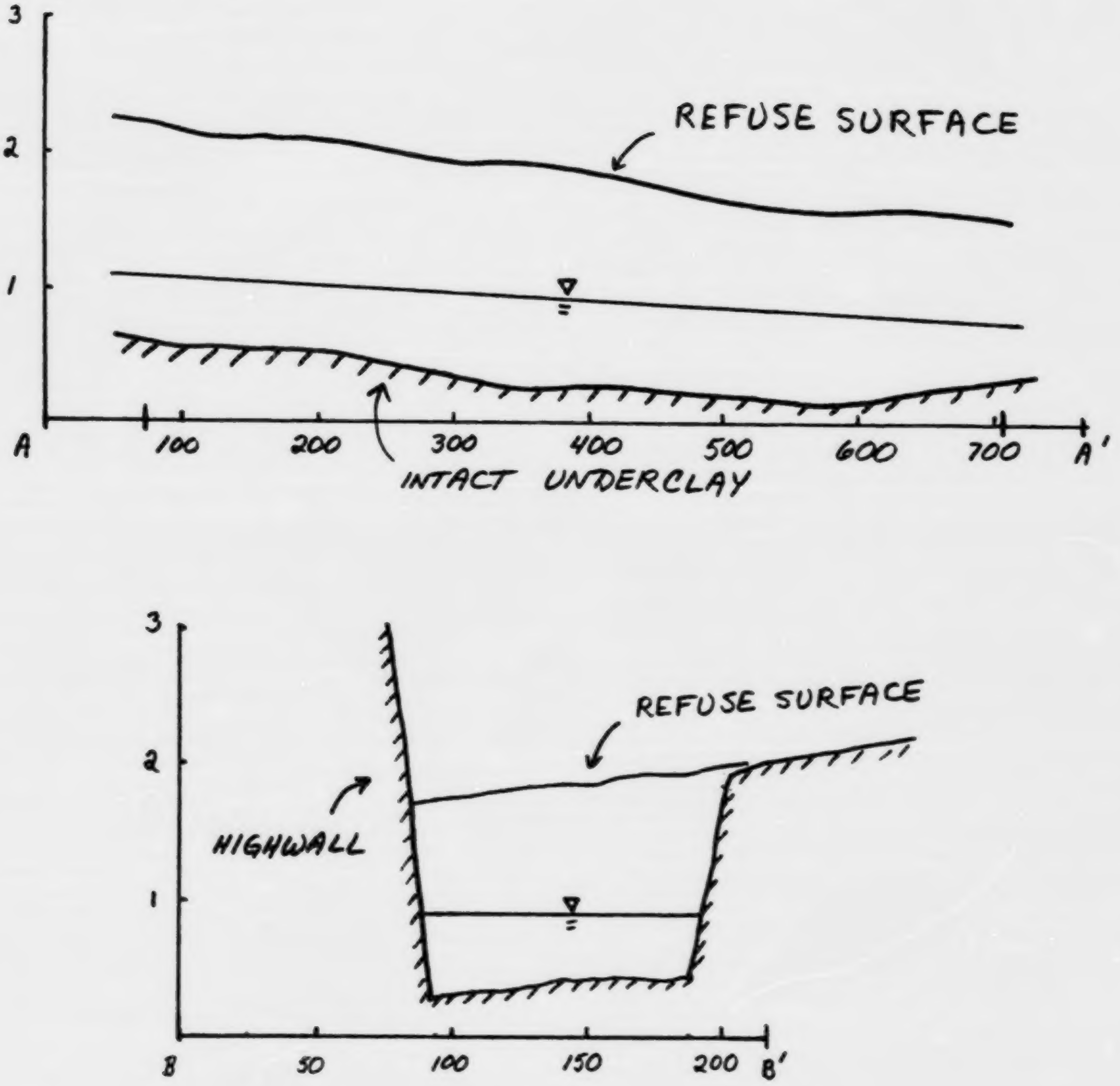
CASE HISTORY : COAL REFUSE


● Water well
× Gas probes
A
A'
B
B'
1
2
3
100 m

PLAN VIEW


3
2
1
REFUSE SURFACE
A
100
200
300
400
500
600
700
A'
INTACT UNDERCLAY
3
2
1
REFUSE SURFACE
HIGHWALL
B
50
100
150
200
B'

CROSS - SECTIONS

AVAILABLE DATA (IDEALIZED)

AREA	2000 M^2
THICKNESS	
TOTAL	1.5 M
SATURATED	0.5 M
DENSITY	1,500 KG/M^3
PRECIPITATION	100 CM/YR
DISCHARGE	
RUNOFF	$1x10^6$ L/YR
SEEPAGE	$1x10^6$ L/YR
SULFUR CONTENT	
TOTAL	7.0 %
PYRITIC	5.0 %
SULFATE	1.0 %
OXYGENATED ZONE	30 CM

WATER QUALITY (IDEALIZED)

	SEEP	RUNOFF
pH	2.0	1.9
Acidity, mg/L*	13,000	20,000
Iron, mg/L	3,500	5,000
Sulfate, mg/L	14,000	19,000

ALLOWABLE DISCHARGE W.Q.

pH	6.0 - 9.0
Acidity	0 (net alkalinity)
Iron	3 mg/L

---

* As $CaCO_3$ equivalent. Divide by 50 to convert to meq/L or mg/L as H^+

REACTION STOICHIOMETRY

$FeS_2 + 7/2\ O_2 + H_2O = Fe^{2+} + 2SO_4^{2-} + 2H^+$

$Fe^{2+} + H^+ + 1/4\ O_2 = Fe^{3+} + 1/2\ H_2O$

$Fe^{3+} + 3H_2O = Fe(OH)_3(s) + 3H^+$

NET:

$FeS_2 + 15/4\ O_2 + 7/2\ H_2O = Fe(OH)_3(s) + 2SO_4^{2-} + 4H^+$

CONVENTIONAL TREATMENT

SET UP A PLANT

- INDICATE UNIT OPERATIONS
- DO CHEMICAL COSTS
- ESTIMATE CAPITAL AND OPERATING COSTS

CONSIDER

- SITING
- SLUDGE STORAGE AND DISPOSAL
- DEGREE OF MECHANIZATION

INFORMATION NEEDED

1. FAILING ANY COST PROJECTIONS OF OUR OWN, TRY THE FOLLOWING

   HYDRATED LIME \$0.10/KG

2. DESIGN CRITERIA

   - REFUSE PRODUCES 33000 KG ACIDITY/YR AS $CaCO_3$
   - TOTAL DISCHARGE 2×10^6 L/YR
   - SUFFICIENT STORED AND POTENTIAL ACID FOR 25 YEARS
   - LIME USAGE WILL BE 84 KG LIME FOR 100 KG ACIDITY

THE CONTROL AND TREATMENT OF AMD FROM AN ABANDONED OPEN PIT AND WASTE ROCK DUMP

-A Workshop Case Study-

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Dr. A.M. Robertson
Steffen, Robertson and Kirsten
Consulting Engineers
Vancouver, B.C.

You are a consultant hired by a government agency to solve an acid mine drainage problem at an abandoned metal mine in British Columbia. The acid mine drainage has been discharging from an open pit and waste rock dump to the environment for about 10 years and has resulted in the loss of important salmon habitat in a downstream river. Impacts occur primarily from copper and aluminum, which are at concentrations above water quality criteria. The mine has never been reclaimed.

You have been asked to review the technologies available to reduce the acid mine drainage and recommend an abatement plan for the mine. This plan should be effective in reducing the acid mine drainage to 5% or less of the current loading to the environment for a minimum of 50 years. The plan should be effective, but must be achieved at the lowest cost possible given limited financial resources available. Details for the site and a list of technologies to be considered are provided.

It is recognized by the client that the amount of abatement and the period for which it will be effective differs for the various technologies. Further, the current state of the art is such that there is considerable uncertainty as to what can be achieved with the various technologies. Secure solutions may be prohibitively expensive. The risk of failure associated with the less secure solutions must be assessed with a view to selecting the technology considered to be the most cost effective.

The Site

- located on Vancouver Island on the west coast of British Columbia about 50 km from the ocean at 1500m elevation (see attached location map).
- subject to 2000 mm of precipitation with 7 months of snow. Mean maximum snow depth in April is 400 cm. The mean temperature of the coldest month (December) is -2.75°C.
- access very difficult for 6 months (December to May inclusive).
- maximum loadings of pollutants to the environment occur in May and October, associated with snowmelt and rainfall events respectively.
- acid generation occurs from an open pit of about 4 ha and a wasterock dump of about 3 ha. The open pit is about 5m deep and the waste rock dump is a maximum of 15 m high (see attached plan and elevation view).

- AMD is formed from snowmelt an precipitation percolating through the mine waste. Groundwater flow from upslope through the waste is considered negligable. Surface water flow from the upstream catchment contributes about 40% of the water entering the mine waste.

- AMD from the open pit arises from loose ore and waste left in the pit at abandonment. Only small amounts of AMD occur from the pit walls. The loose material in the open pit amounts to 100,000 tonnes, while the waste rock dump is 300,000 tonnes. The average sulphur level is 3% and neutralization potential is 10 tonnes/ 1000 tonnes of waste. The density of the waste is about 1.8 tonnes/ cubic metre.

- AMD from the open pit flows over the pit floor and enters the waste rock dump along with direct precipitation on the dump. Additional AMD is generated in the waste rock dump. The AMD re-appears as a series of seeps at the toe. These seeps are at the same location year round, but, their quality and flow varies. The total seep flow reaches 200 l/sec in May and 100 l/sec in October. The mean concentrations of important pollutants in the seeps are as follows:

pH	3.0 pH units
acidity	200 mg/l as CaCO3
D. Cu	10 mg/l
D. Al	10 mg/l
D. Fe	3 mg/l

- dump is very porous and there are no seeps on the slope of the dump except at the toe.

- AMD from the open pit and waste rock dump is considered to be the sole source of metals to the environment.

- no power at the site. The nearest connection to the electrical grid is 8 km distant.

Construction Material Availability

- approximately 44,000 cubic metres of till have been located about 3 km from the site. Silt and clay sizes in the till are only 15 to 20%. The permeability of the till is relatively high at 1×10^{-5} cm/sec.

- no natural carbonate bearing rock occurs in the area.

- limited amounts of organic soils and peat are present near the site.

- AMD flows into a creek which passes through a natural marsh some 0.5 km downstream of the mine site. The marsh is about 4 ha and is entirely composed of aquatic grasses. The influent concentrations are essentially the same as the seep quality. The marsh is at about 1200 m elevation and exhibits growth in June to September. Plants appear to be dormant at other times. Peak flows in the creek are about 250 l/sec in May and 150 l/sec in October.

- two small lakes with low fisheries significance are within 1.5 km of the minesite. One has a water volume of 125,000 cubic metres while the other is 200,000 cubic metres.

- with the exception of till, all construction materials must be imported to the site from greater than 50 km distance.

Abatement Options

Consider the following 10 options to reduce the acid mine drainage from the mine:

1) compacted till covers
2) synthetic membrane covers
3) surface applications of limestone
4) diversion ditches
5) alkaline trenches
6) removal of waste to lakes
7) topsoil and revegetation
8) bactericides
9) chemical treatment
10) wetland treatment

Assignment

In your group, develop an AMD abatement plan using the option assigned at the workshop session. This plan will use the option concept, but, the details and implementation will be decided by the group under the direction of the group leader. After the details of the plan have been developed, assess the plan using the scoring system provided. Reach a concensus on each of the scoring items and the group leader record the value. After scoring the plan, estimate the capital and the operating costs over the next 50 years (assume 0% inflation) using the unit costs provided and any other knowledge available. Where cost data are not provided, and no one in the group has experience, take a guess! When these two tasks are completed, the group leader totals the ranking points and calculates the cost/benefit ratio at the bottom of the sheet:

$$\text{cost/benefit} = \frac{\text{Preliminary total cost}}{\text{Total ranking points}}$$

The group leaders hand the sheets to the workshop facilitators as soon as completed. You have about 25 minutes to complete this exercise. (It may be efficient to divide the ranking and costing tasks among members of the group to save time depending on the complexity of your option and plan.)

Following completion of your group task, individually consider the 10 options listed above and list the 3 on the attached sheet that you feel would have the highest point ranking and the lowest cost/benefit ratio ranking (ie. the best options according to the two criteria). Also hand this sheet to the workshop facilitators. You have about 10 minutes.

CONVERSION FACTORS

Length
1 km = 0.62 miles
1 m = 3.28 feet
1 cm = 0.39 inches
1 mm = 0.039 inches

Area
1 ha = 2.47 acres
1 m2 = 10.8 feet2

Volume
1 l = 0.035 feet3
= 0.26 U.S. gal
1 m3 = 35.3 feet3
= 264 U.S. gal

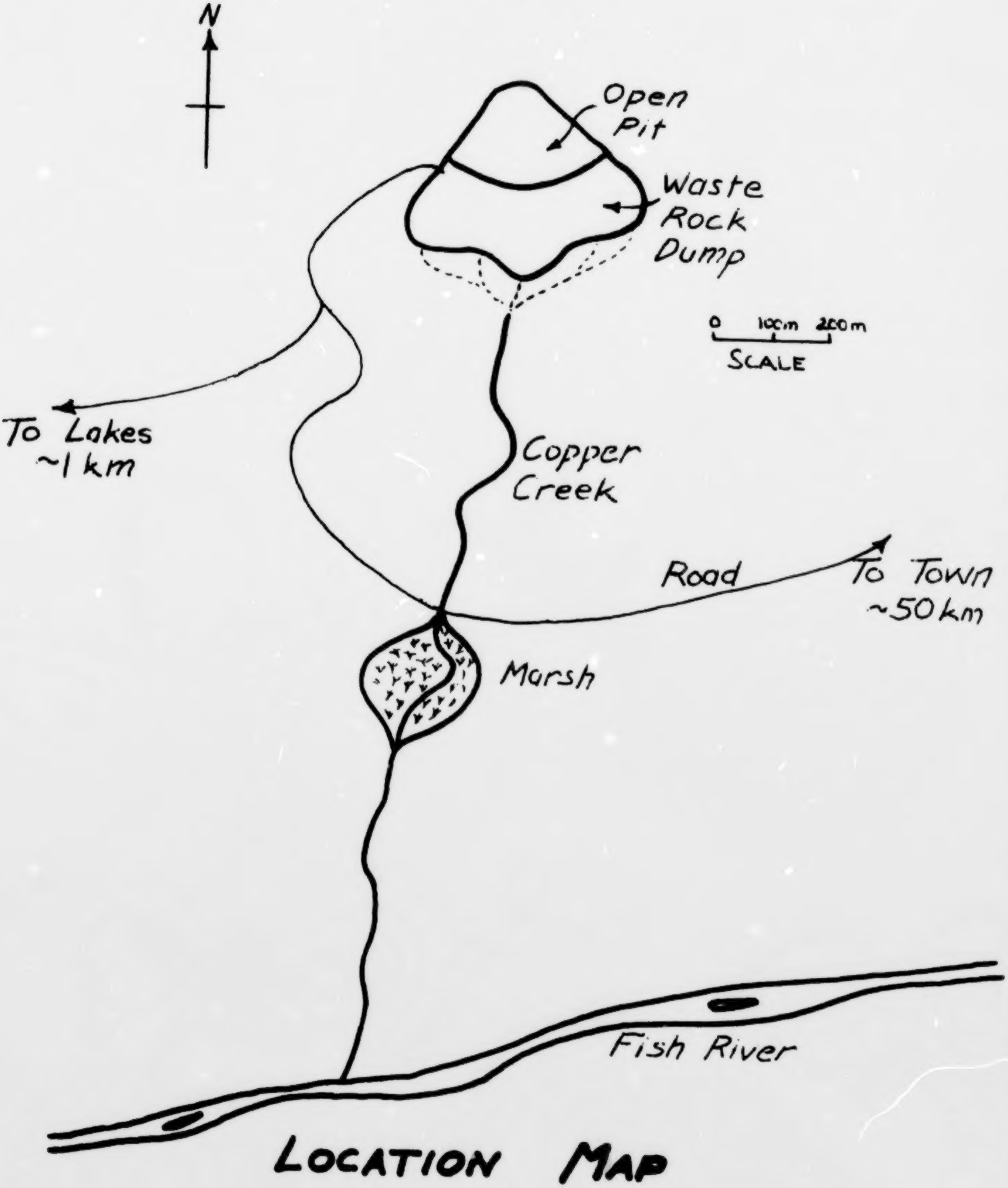
Mass
1 tonne = 1.1 tons
1 kg = 2.2 pounds

Density
1 t/m3 = 0.84 tons/yd3

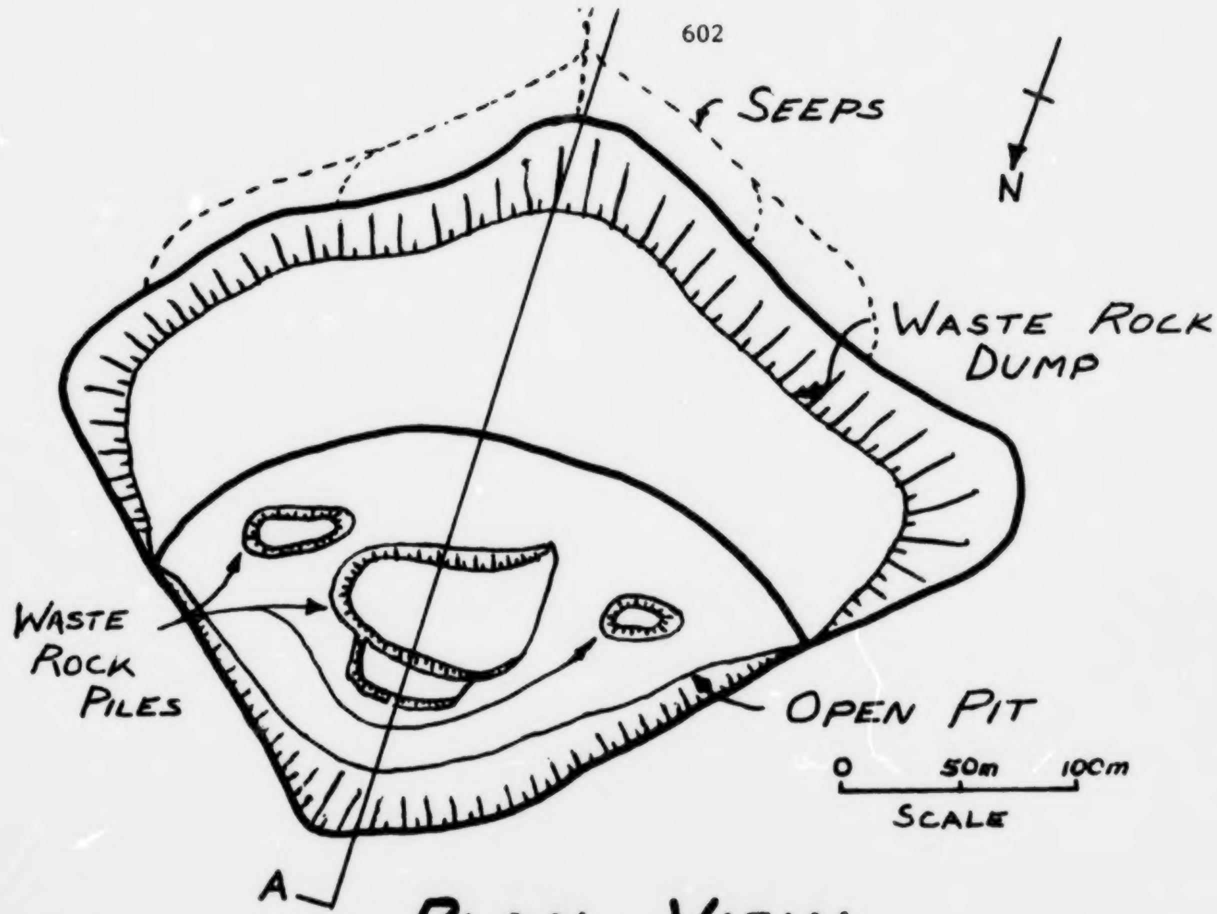
Velocity
1 cm/sec = 0.033 feet/sec

Flow
1 l/sec = 15.8 U.S. gal/min

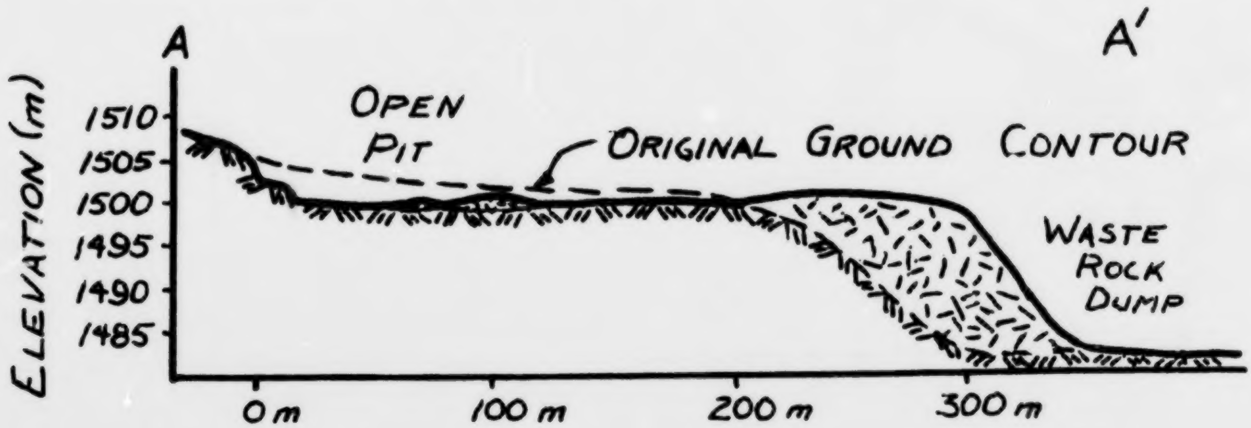
Temperature
°F = 9/5 °C + 32


N
Open Pit
Waste Rock Dump
0
100m
200m
SCALE
To Lakes ~1 km
Copper Creek
Road
To Town ~50 km
Marsh
Fish River

LOCATION MAP


SEEPS
N
WASTE ROCK DUMP
WASTE ROCK PILES
OPEN PIT
0
50m
100m
SCALE
A

PLAN VIEW


A
A'
ELEVATION (m)
1510
1505
1500
1495
1490
1485
OPEN PIT
ORIGINAL GROUND CONTOUR
WASTE ROCK DUMP
0m
100 m
200 m
300 m

SECTION A-A'

PLAN AND ELEVATION VIEWS

UNIT COSTS FOR AMD ABATEMENT PLANS

Waste collection and placement on-site	$0.60/m3
Drainage ditching-loose rock	$10/m
-hard rock	$40/m
Site grading	$5000/ha
Revegetation (mulching, seeding, fertilizing)	$6000/ha
Rip-rap	$20/m3
Till -excavating, loading and dumping	$1.75/m3
-transporting	$0.50/m3
-spreading and compacting	$0.75/m3
Synthetic liner	$18/m2
Limestone-delivered to site	$26/m3
-spreading and scarifying	$0.75/m3
-scarifying alone	$700/ha
Mixed alkaline materials (sodium carbonate lime, limestone)	$34/m3
Waste excavation-to trucks	$2.75/m3
-to piles on-site	$0.75/m3
Lime application to site	$30/m3
Bactericides-spray on-site	$10,000/ha
-pellets on-site	$35,000/ha
Chemical treatment-power	$10,000/km
-chemicals	$41,000/yr
-storage buildings, piping and tanks	$36,000
-neutralization tanks, piping, and instrumentation	$40,000
Wetlands development, mulching, and planting	$15,000/ha

ACID ABATEMENT ALTERNATIVES RANKING SYSTEM

	Consideration	Description	Scale					Weighting Factor	Point Total
			Low		Medium		High		
1.	Practicality	Technical feasibility	1	2	3	4	5	x4	
		years	1	10	50	100	>100		
2.	Durability	Life expectancy	1	2	3	4	5	x2	
			High		Medium		Low		
3.	Security/Risk	Risk of failure	1	2	3	4	5	x4	
4.	Environmental Impacts	Adverse impacts due to construction or operation							
			High		Medium		Low		
		(a) Construct	1	2	3	4	5	x1	
		(b) Operation	1	2	3	4	5	x1	
			Maximum				Minimum		
5.	Construction	a) Complexity, magnitude	1	2	3	4	5	x2	
		months	>6	4-6	2-4	1-2	<1		
		b) Duration of construction	1	2	3	4	5	x1	
			Wkly	Month	Quat.	Ann.	>Ann.		
6.	Inspection and Maintenance	a) Frequency of inspection	1	2	3	4	5	x1	
			High	Mod.	Low	V.Low	None		
		b) Level of maintenance	1	2	3	4	5	x5	
			<75	75-90	90-95	95-100 less secure	95-100 more secure		
7.	Effectiveness	Reduction of contamination	0	2	3	4	5	x10	

TOTAL RANKING POINTS

Cost/Benefit = $\frac{\text{Preliminary Total Estimate}}{\text{Total Ranking Points}}$ =

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Review of the National Uranium Tailings Programs Research into Acid Generation

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In 1981 the National Technical Planning Group on Uranium Tailings Research concluded that there was not enough scientific data to predict confidently the long term effect of uranium tailings on the environment (1). The Government of Canada responded to this recommendation by announcing, in 1982, the formation of the National Uranium Tailings Program (NUTP). This is a five year research program due to end in March 1987 that will have received a total of about $8.5 million in funds. While the principle objectives were to study the ways to predict the effects on the environment of uranium mill tailings over the long term with particular emphasis on the movement of radionuclides, it was also essential to understand the process of acid generation.

History

There are two sites with acid generating uranium tailings in Canada (2). The principle one is at Elliot Lake in Ontario with a smaller site at Laredo in northern Saskatchewan.

Laredo Uranium Mines Ltd. operated a custom mill from 1957 to 1960, using an acid leach circuit. Approximately 360,000 tonnes of tailings were deposited at pH 2 directly into Nero Lake. These tailings are pyritic and acid generating although they have now reached a steady state with their surroundings. The tailings themselves are partly underwater, having been dumped at the edge of a small lake. No vegetation grows on the exposed tailings and some of this material has blown into the surrounding area destroying the adjacent growth. The lake does have a minimal community, mostly consisting of a moss carpet. A precipitate plume, high in radionuclides, extends through a berm between the small lake and the much larger Beaverlodge Lake.

Elliot Lake has many uranium tailings sites, containing a total of about 100 million tons. These tailings have been generated from a reef-type ore that is similar to the conglomerate ores of Witwatersrand in South Africa. They contain pyrite and phyrotite in appreciable quantities (3 to 5 percent). All of the mills in the area have used strong acid leach. These conditions destroy iron sulphide minerals such as phyrotite and marcasite but not pyrite.

Before the early '60's environmental issues did not receive a great deal of attention. At that time, monitoring of the Serpent River system, a complex basin of lakes and rivers draining the Elliot Lake region, revealed that the pH had dropped to a point of concern. Measurements taken over the next few years revealed that significant portions of this surface water system had ceased to contain the normal aquatic communities. Corrective action was taken by the mining companies. Initially this was neutralization of the tailings, a more judicious use of water and, ultimately, the removal of radium as a co-precipitate with barium sulphate. Additionally, to reduce seepage losses, new dams were carefully engineered with either clay or plastic cores. Seepage from older facilities was collected and neutralized. As a result of these efforts the Serpent River system is now virtually restored, and the area is being restocked with fish (3).

No remedial action has taken place at Laredo however. This is a very isolated location, with a small amount of material that has little impact except on the immediate environment. It will be difficult to decide what action, if any, should be taken at this location.

The Role of Acid Generation in the NUTP

Primarily the NUTP staff were interested in calculating radiological doses to human beings. The major pathway for that dose is through the surface water system, either by direct uptake or via the consumption of fish. Radionuclides such as uranium, radium and thorium dissolve within the tailings matrix into the pore water and eventually emerge as seepage to enter the surface water system. To understand the dissolution chemistry of these elements it was essential to calculate the pH conditions. As the concern was for the long term (hundreds of years) it was also necessary to understand the evolution of acidity and thus the change in pH over long periods of time. For these reasons the staff initiated six projects directly related to pyrite oxidation and parts of many other projects contributed to our overall understanding (4-9).

The Oxidation of Sulphur Containing Minerals

Pyrite is the most widespread of the sulphide minerals contained in uranium tailings, but it is not the only possibility. In the following discussions FeS_2 is used to represent the pyritic agglomerates under study (equation 1).

$$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2Fe^{2+} + 4SO_4^{2-} + 4H^+ \quad ....1$$

After release in the neutralized state the tailings are exposed to atmospheric oxygen which chemically attacks the iron sulphide. This is a slow reaction, but with time the tailings approach conditions that allow for enhanced bacterial activity. Thiobacillus ferroxidans is the principal bacterium which oxidizes the ferrous iron produced in the initial reaction, to ferric (equation 2).

$$4Fe^{2+} + 4H^{+} + O_2 \rightarrow 4Fe^{3+} + 2H_2O \quad ....2$$

In making this change it can gain from a two electron transfer through a cytochrome intermediate that releases biologically useful energy (equation 3a & b).

$$2Fe^{2+} \rightarrow 2Fe^{3+} + 2e^{-} \quad ....3a$$

$$4H^{+} + O_2 + 4e^{-} \rightarrow 2H_2O \quad ....3b$$

The ferric iron then produced reacts with more sulphide to give new ferrous iron and a further incease in acidity. This process continues until the tailings are at the optimum pH (2-3) for bacterial activity (equation 4).

$$14Fe^{3+} + FeS_2 + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^{+} \quad ....4$$

For the reaction to continue as a bacterially mediated process there has to be 40 to 50 kJ/mol (this is the adenosine triphosphate equivalent energy and represents a biologically useful quantity of energy). The free energy generated as a result of the ferrous iron oxidation is given by equation 5.

$$\Delta G^{\circ}_{pH} = -88.6 + 11.4 \text{ pH} \quad ....5$$

By substitution it can be shown that when the pH reaches approximately 4.3, the available energy drops below 40 kJ/mol and ceases to be biologically useful. Thus, above pH 4, ferric iron is lost as the precipitated hydroxide (equation 6). At these pHs the pyrite surface shows pitting, which indicates direct oxidation of the sulphide by thiobacillus species. This attack however is not occurring in such a manner as to be influencing the rate determining step.

$$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 + 3H^{+} \quad ....6$$

Under more acidic conditions the rate determining step is shown by equation 2. The most critical factor is the availability of oxygen. The data indicate that the critical oxygen concentration is approximately 5% of the oxygen concentration at saturation (that is 0.5 milligrams per litre). At this level a bacterially mediated reaction occurs but the production rate of ferric iron and hence hydrogen ion is limited. These oxygen levels can occur down to a depth of 1 to 2 metres in Canadian tailings.

These reactions are also temperature dependent. The limited data show that the temperatures found at 2 metres approximate the average ambient air temperature and vary little through the year. At one metre there is some variation, but this is considerably damped when compared to the surface fluctuations. For long-term practical purposes the temperature can be considered as virtually constant.

Role of Modelling

The NUTP objective when stated in general terms, is to supply enough information and an assessment methodology to help choose site-specific tailings disposal options. As the NUTP was seeking a means for assessing the long-term implications of these disposal options, developing a predictive mathematical model was a key component.

The model developed to date (called UTAP for Uranium Tailings Assessment Program) consists of a system modelling framework that integrates component models to predict the long-term effects of uranium mill tailings and allows for an assessment of the cost effectiveness of various disposal options (10). The component modules that form part of this framework describe the source term, aquatic, atmospheric and terrestrial transport of contaminants, biological uptake, and finally dose to humans. In each of these component modules, the complex chemical, physical and biological behaviour of the contaminant source and exposure pathways are approximated by mathematical models. As many of these parameters are uncertain or variable a probabilistic analysis approach was selected to allow for the incorporation of these uncertainties. In a probabilistic analysis model input parameters are assigned distributions of values rather than single values as specified in the traditional deterministic method.

As part of a study to estimate the rate and quantity of acid generated by the bacterially assisted oxidation of sulphide minerals, a computer model (RATAP for Reactive Acid Tailings Assessment Program) was developed (11). This model analyses the effects of the oxidation processes and variables on the rate and quantity of acid generated. The RATAP model is divided into two parts; control module and component modules. The control module acts as an executive and provides the infrastructure required for the probabilistic analysis of the component modules. The six component modules, in order of operation are: initial inventory, kinetics, oxygen transport, pyrite oxidation, solute transport and aqueous solute speciation. The concept adopted for modelling the tailings is shown in figure 1. In the unsaturated zone and capillary fringe 1 to 20 layers are employed, whereas in the saturated zone 1 or 2 layers are used. The exact number of layers are specified in the input file. The call sequence of these modules over time for the various layers is shown in figure 2.

The first module, initial inventory, calculates for each layer the initial solids composition, soil temperature, water content and air content. The kinetics module calculates rate information at each time step for the various layers. The oxygen consumption rate calculated is then employed in the oxygen transport model and results in a calculated oxygen profile. The following three modules all run in sequence at time t for each layer. The pyrite module results in an overall rate of pyrite oxidation. The solute transport and aqueous speciation modules use this information to calculate the acid flux and pH respectively. These processes are repeated for time period t + 1 and so on.

Total acidity for the sake of this exercise is taken as equivalent to the CO_2-acidity titration end-point. Hence total acidity is defined as the amount of base which would be required to neutralize free protons, all species of aluminum and iron (III) and carbonic acid to this end point. To achieve this the model requires approximately 173 coefficients which must be entered as a nominal value or as a probability distribution. Much of this information can be obtained from existing literature data. However, a number of coefficients are site-specific.

Nominal values of the coefficients for the initial characteristics were selected for an Elliot Lake site. The model was then run for a 20 year simulation and the predicted results were compared to measured field data. In general there was reasonable to good agreement with the field observations, thus providing some confidence in the predictive capability of this model. These deterministic simulations on a representative sulphidic tailings example indicated that the rate of pyrite oxidation exhibits a strong dependence on temperature, bacterial population density and the parameters controlling oxygen diffusion. They also indicate that bacterially assisted oxidation predominates over chemical oxidation by approximately 6 times at the acidic pH's and between temperatures of 4° to 15°C.

It has not been the intention in this paper to report on the detailed mathematics used in RATAP. These are fully given elsewhere (11). Suffice to say that RATAP is based on the chemical principals given above. It is more appropriate to consider the use to which output from RATAP can be put in resolving tailings management issues.

Practical Considerations

Four key parameters emerge from the NUTP work as being worthy of consideration for acid drainage control schemes. The first parameter, temperature, is largely outside our control. Although the addition of cover material will serve to dampen temperature fluctuations within the tailings mass there will not be a great deal of change in the average value. The second parameter, bacterial population density, can be controlled by the addition of various reagents. However, the effect of these materials is relatively short lived even with time-release formulations. Thus their influence could only be maintained by regular additions; not a satisfactory practical solution. The remaining two parameters, control of oxygen diffusion and reduction of water movement, can be controlled and have therefore been subject to more investigation.

An early NUTP project investigated the use of synthetic barriers for both oxygen and flow control (12). Although it was possible to select the most effective barriers there remained serious questions about the long-term stability of these materials.

Attention was then given to natural cover material that was expected to have long term stability (13). Vegetative cover while inexpensive and having a number of advantages, appears to have little or no effect on the sub-surface oxidation of pyrite and would be difficult to maintain on phytotoxic tailings. Soil cover is more expensive than simple vegetation (which would probably be required to prevent erosion anyway) but does effect the oxygen diffusion rate. If properly engineered it can also reduce infiltration rates. Waste rock (or rip rap) cover is even more expensive especially if some suitable form of mine waste is not available. It actually increases the infiltration rate and is not as an effective oxygen barrier as soil. Water cover is an excellent inhibitor of oxidation rate and the water itself is inexpensive. However, the stability of the structures required to maintain the water level is open to question. Should these structures be lost then the entire tailings would be exposed with subsequent massive oxidation and environmental damage. Using, where available, a natural and stable deep lake for disposal has not been politically acceptable. For the rare occasions when tailings have been deposited underwater the results to date are very promising. Questions,however,still remain about a number of long-term issues.

The bulk of Canada's two billion tons of tailings are deposited on surface. It would therefore appear that the soil (or equivalent) cover option will be the most effective. Eventually, whether deliberately seeded or naturally colonized, the area will be vegetated, with the ultimate growth being forest. Almost certainly other techniques such as surface contouring and the use of rip rap in high erosion areas will have to be included.

Although soil cover represents a generic solution, there are many details that are still unknown. For example, it has been shown that roots of trees growing on a shallow surface cover do not penetrate the harsh conditions of the tailings themselves. When the trees reach a certain weight the prevailing wind conditions are sufficient to topple them **over,thus making large breaches in the surface cover.** Little work has been done to establish what types or combination of types of soil cover are best at reducing the oxygen diffusion rate. Soil cover alone permits salt migration. Can this be avoided by some form of capillary break? What is the long-term stability of the structures holding the tailings and their covers in place? The evidence suggests that the tailings retain high levels of moisture for long periods and therefore remain reasonably fluid. Any breach in the retaining structure would allow tailings to flow into uncontrolled areas.

Probabilistic assessment, using RATAP, indicates the most active period for sulphur bearing uranium tailings is about 10 years. After this time the main flux has decreased to a quasi-steady state. This would suggest that active tailings management should continue for at least a decade after shutdown of a tailings area.

Conclusions

The National Uranium Tailings Program, although not primarily concerned with pyrite oxidation and subsequent acid generation, has nevertheless had to address many of the questions that face other heavy metal tailings in this respect. To that end it has undertaken considerable amount of research to establish the key factors and controlling mechanisms. These have been outlined in a series of reports from which some of the key items have been summarized in this paper. Specifically, a probabilistic predictive mathematical model (RATAP) has been generated and, to a reasonable extent, validated. This allows for the calculation of anticipated concentrations of acidity and pH levels with time, for sulphide bearing uranium tailings. The model is built in such a way that it can be adapted for use with other non-uranium tailings. Unfortunately, at this time, RATAP is not user friendly, something which may be remedied in the near future. In addition, a larger more complex model (UTAP), which is user friendly is available to calculate the long-term behaviour of a variety of elements and ions, including non-radionuclides such as nickel and arsenic. It is intended to incorporate all of the appropriately useful work done by NUTP on modelling, fundamental science and disposal techonology into the Reactive Acid Tailings Program (RATS) (14).

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